2006

Coastal marsh restoration using terraces: effects on waterbird habitat in Louisiana's Chenier Plain

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COASTAL MARSH RESTORATION USING TERRACES: EFFECTS ON WATERBIRD HABITAT IN LOUSIANA'S CHENIER PLAIN

A Thesis

Submitted to the Graduate Faculty of the
Louisiana State University and
Agricultural and Mechanical College
In partial fulfillment of the
requirements for the degree of
Master of Science

in

The School of Renewable Natural Resources

by
Jessica O'Connell
B.A., Sonoma State University, 2001
December 2006

AKNOWLEDGEMENTS

This project was funded by the USFWS Gulf Coast Joint Venture and Ducks Unlimited Inc., project number LA-86-1. I thank the staff of Rockefeller State Wildlife Refuge (SWR) for providing housing, logistical support, and the occasional rescue. I thank Rockefeller SWR, Sabine National Wildlife Refuge, The National Audubon Society, Vermilion Corporation, Sweet Lake Gas and Oil Company, and Miami Corporation for allowing me access to their land. I thank everyone who helped me in the field, especially Christina Leglue and Melissa Miller for their consistent hard work and affability in the face of unpredictable circumstances, but also Marie Perkins, Chris Cannaday, Bryan Gossman, Jessica Hanks, Jennifer DiMiceli, and Whitney Broussard for coming out and assisting me when I needed them.

I would like to thank the members of my graduate committee, especially my major professor Dr. J. Andrew Nyman for his continual help, guidance and support, and Drs. Alan Afton, Irv Mendelssohn, and J. V. Remsen for consistently providing sound feedback and advice.

Finally, I would like to thank my parents, Philip and Stephanie O'Connell for setting me on this path and encouraging me along the way.

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ABSTRACT

Terracing is a novel technique used to combat coastal marsh loss in Louisiana and Texas. Terraces are assumed to slow marsh erosion, decrease pond depth, and encourage vegetation expansion. Terraced ponds have never been evaluated as habitat for waterbirds, which heavily depend on Louisiana's coastal marshes. From April 2005 to April 2006, I monitored waterbird species richness and density through time to estimate effects that terracing has on habitat quality. Water quality (turbidity, salinity, conductivity, water temperature, and water depth) also was measured. Submerged aquatic vegetation (SAV) biomass and nekton density were measured from April 2005 to September 2005. I monitored paired terraced and unterraced ponds in three sites within Louisiana's Chenier Plain. Observations and samples were taken in two microhabitat types within ponds: marsh edge and open water.

Terracing ponds increased the proportion of marsh edge, but did not alter water quality variables measured. SAV and nekton were denser at the marsh edge than in open water, but did not differ significantly when compared at the whole-pond level between pond types. Waterbirds also were denser at the marsh edge. Waterbird density was consistently greater in terraced ponds. Waterbird species richness was greater in terraced ponds in winter and during spring and summer was generally greater in terraced ponds. Additionally, bird density in ponds varied by foraging guild. During spring and summer, aerialists, shorebirds, and dabbling foragers were consistently denser in terraced ponds. Wading forager densities varied in ponds with time, but were generally denser in terraced ponds. Diving foragers were not dense and did not differ between pond types. During

winter, only dabbling and wading foragers were significantly denser in terraced ponds, but these two guilds represented 83% of birds observed. Other foraging group densities did not differ between pond types. Several species of conservation concern were observed. Trends in density for most species of concern were similar to those seen for the foraging guild in which that species was classified. Marsh edge is a biologically prolific habitat. The amount of edge necessary to achieve pond level effects for nekton and SAV has not been evaluated.

CHAPTER 1: PROJECT OVERVIEW

The overall objective of this study was to investigate the efficacy of marsh restoration using terraces at improving waterbird habitat quality in Louisiana's Chenier Plain. This project consisted of three studies: a pilot study testing the ability of photosensitive cameras to conduct bird surveys over large areas (Nov 2004 to Jan 2005), a pre-Hurricane Rita study using observers (April to Sept 2005), and a post-Hurricane Rita study using observers (Jan to April 2006).

The post-hurricane change in survey methods (Table 1) was required because of the extensive amount of damage to infrastructure and development in the Chenier Plain following Hurricane Rita. A two-month gap in surveys was required because the region was inaccessible directly following the hurricane. After surveys were resumed, a shortening of their intensity and length was necessitated because of a lack of housing in the area. Methods differed between the pre and post hurricane studies, and thus precluded combing of their results into one study.

Table 1. Differences in methods between pre and post Hurricane Rita studies.

Pre-hurricane study	*Post-hurricane Study*	*Justification for change*
Birds surveyed at dawn	Birds surveyed at various times during midday	Allowed more sites to be surveyed/day. Prevented conflicts with waterfowl hunters.
Nekton and submerged aquatic vegetation (SAV) samples collected	Nekton and SAV not collected	Allowed bird surveys at more sites each day
Birds surveyed over a 90 min period at each plot with a settling period before observations began	Birds counted quickly (no settling period) and only once at each plot	Allowed bird surveys at more sites each day
One plot/pond sampled during each survey	Multiple plots counted each survey	Surveys at each plot were shortened, thus there was time to sample multiple plots/pond.
Unequal plot sizes (4 to 13 hectares)	Roughly equal smaller plot sizes (3 to 5 hectares)	In the pre-study, unequal plot size was a result of adding sites later in the study. For the post study, plot size is equal because sites are consistent through the study, and small because quick counts were easier on small plots.
Bird behavior and microhabitat usage data recorded consistently	Bird behavior and microhabitat usage data recorded when possible	There was no post-study settling period before surveys. If birds were disturbed by observers, collecting behavior and microhabitat data was not possible.

CHAPTER 2: WATERBIRD, NEKTON, AND SAV IN TERRACED VS. UNTERRACED PONDS DURING SPRING AND SUMMER

Coastal marshes are highly productive and constitute valuable wildlife habitat. Wetlands in the United States are vastly reduced from historic ranges (Dahl 1990). Louisiana contains the largest area of remaining wetlands in the continental United States (NOAA 1991). These marshes provide critical habitat for waterbirds, for which many populations are suffering regional declines concomitantly with habitat loss. For example, a significant proportion of the continental population of wading and seabirds (25% or more of the total US population for 10 different species) use Louisiana's coastal marshes. Additionally, they contain more nesting colonies of seabirds and wading birds than any other state in the southeast (Keller et al. 1984, Martin and Lester 1990). Further, 13 species classified as species of high concern by the Waterbird Conservation Council (Kushlan et al. 2002) are regularly seen in brackish marshes along the Gulf Coast (Black Skimmer, Least Tern, Little Blue Heron, Snowy Egret, Tricolored Heron, Gull-billed Tern, Roseate Tern, Pied-billed Grebe, Purple Gallinule, American Bittern, and King Rail)[*]. An additional 14 species of moderate concern also occur (American White Pelican, Forster's Tern, Anhinga, Neotropic Cormorant, Reddish Egret, Roseate Spoonbill, White Ibis, Black-crowned Night-Heron, Eared Grebe, Royal Tern, Clapper Rail, Virginia Rail, Common Moorhen, Common Loon). Located at the confluence of the Mississippi and Central flyways (Bellrose 1980), coastal Louisiana provides critical stopover and wintering habitat for 20% or more of the continental population of 14 species of waterfowl (Michot 1996, Esslinger

[*] Please see Appendix A for scientific names of all species used in text

and Wilson 2001). However, wetlands in coastal Louisiana have been in rapid decline. Louisiana's coastal wetlands comprise 40% of those found in the US, and have sustained 80% of coastal wetland loss from 1950 to 1994 (Boesch et al. 1994). Most recent loss rates, calculated for the 1983 to 1990 period, are 65.6 km^2/year for the entire coastal plain (Britsch and Dunbar 1993).

Coastal land loss in Louisiana occurs for a variety of reasons. The vast majority of this loss results from conversion of marsh to shallow open water. Historically, marsh loss was part of a natural cycle of land gain in areas receiving high sediment loads from riverine or storm inputs, or in those with high plant peat production, balanced by land loss due to soil erosion and compaction (Neill and Deegan 1986, Wells and Coleman 1987, Mitsch and Gosselink 2000). Currently marsh loss far outweighs marsh gains in most coastal regions within Louisiana (Barras et al. 2003).

Coastal marshes in Louisiana can be grossly divided into two zones: the deltaic zone in the east, containing primary sediments from active and inactive Mississippi River deltas, and the Chenier Plain in the west, containing riverine sediments that have been secondarily reworked by Gulf Coast wave action and then redeposited back onto the land. The Chenier Plain of Louisiana traditionally has been thought to be more stable than the deltaic plain (Barras et al. 1994), having only 20% of total wetland loss from 1978-1990 as opposed to 80% in the deltaic zone, but marsh loss is still significant. From 1978 to 2000, loss rates in the Chenier Plain were 16.3 km^2/year (Barras et al. 2003). Unlike delta marshes, direct sediment delivery from rivers and bayous is limited in the Chenier Plain to a few localized areas. To maintain marshes, the Chenier Plain relies on mineral supplements from sporadic over-wash events during large storms and on autochthonous organic peat production (Foret 2001). Organic and mineral accretion are interrelated. Mineral supplements have been noted to increase soil fertility, and thus promote increases in plant production (Delaune et al. 1979). Similarly, areas with low fertility because of limited mineral supplements often show compensation via increases in belowground plant biomass production, resulting in accretion rates similar to more fertile soils (Foret 2001). Thus, land accretion in the Chenier Plain relies on background belowground biomass production, interspersed by intermittent mineral accumulation and increased above ground biomass production following sediment inputs from storm over-wash (Foret 2001).

Prior to 1956, much of the Chenier Plain was uninterrupted emergent marsh (Barras et al. 1994). Marsh loss in this region resulted from two main causes: shoreline retreat related to wave energy, and sediment starvation in areas far from riverine sources (Byrnes et al. 1995). Interior marsh breakup continues after an initial vegetation die-off in hot spots. Initial causes of interior marsh breakup in the Chenier Plain are not well understood. A variety of hypotheses have been set forth, including hydrologic alterations such as saltwater intrusion from canal dredging (Baumann and Turner 1990, Turner and Rao 1990, Gammill et al. 2001), geosyncline downwarping due to groundwater or oil and gas removal (Gosselink 1979), prolonged flooding and increased water depths due to various management projects (Gammill et al. 2001), toxic effects from industry runoff (Gosselink 1979), muskrat and nutria eat outs, and ill-timed droughts (Bolduc and Afton 2003). Regardless of causes of initial die off, areas of open water often then spread via soil erosion due to wave energy in larger open-water areas. This phenomenon may be exacerbated by a variety of factors, including global sea level rise and sediment starvation caused by the channelization of the Mississippi and other rivers (Boesch et al. 1994, Turner 1997).

Pond terracing is a novel technique thought to enhance or improve certain functions in degrading marshes. Terracing was developed in response to conversion to open water of marshes in the Chenier Plain (Underwood et al. 1991, Steyer 1993, Rozas and Minello 2001). Terraces are discontinuous, narrow strips of created marsh. They are formed of dredge material stabilized by planting with emergent vegetation such as *Spartina alterniflora* (Underwood et al. 1991, Steyer 1993, Rozas and Minello 2001). Sediment for terrace building usually is taken from the pond bottoms, and is piled using backhoes, creating barrow pits within ponds. Terraces are thought to function by reducing wave energy (Underwood et al. 1991, Boesch et al. 1994) and by dampening the erosive force of water. This is assumed to slow marsh loss and encourage sediment settling. Additionally, water clarity should be increased, resulting in increased production of submerged aquatics. Increased sediment settling additionally may decrease pond depths, increase soil fertility, and provide a more hospitable environment for the expansion of emergent vegetation. Terracing also is thought to improve habitat by increasing the amount of edge (boundary between emergent vegetation and open water) within a pond (Rozas and Minello 2001). Shallow marsh edge frequently has been noted as a highly productive zone for plants, nekton, and invertebrates (Gosselink 1979, Peterson and Turner 1994, Chesney et al. 2000, Minello and Rozas 2002) because it provides a shallow low-energy area where detritus may accumulate, and also because vegetated edges may serve as a nekton nursery and refugia from large aquatic predators. Increasing the proportion of marsh edge has been noted to maximize waterbird density and diversity in marshes in the northern USA and Canada (Weller and Spatcher 1965, Mack and Flake 1980, Kaminski and Prince 1981, Murkin et al. 1982, Fairbairn and Dinsmore 2001) because it potentially increases production of forage items and maximizes habitat interspersion of cover and water. Thus, increasing the proportion of marsh edge should improve habitat quality for wildlife.

Terrace construction recently has become popular in the Chenier Plain of coastal Louisiana. The first terraces were constructed at Sabine NWR in 1990, but the bulk of terrace construction began after 1998 and continues to the present day. The exact number of terracing projects is unknown, but to date, at least 27 have been funded (Table 2). These projects conservatively include approximately 220 km of terraces, affecting a total of 5487 acres of surrounding marsh (Stead and Hill 2004). More projects have probably been initiated than those summarized.

The efficacy of terraces at improving marsh functions can be measured at two scales. First, effects can be compared between areas directly adjacent to terraces edges (restoration condition), and open water habitat far from any edge (unrestored condition). Additionally, although this has rarely been measured, effects should also be evaluated at the whole-pond scale.

Six previous studies have evaluated the effects of terracing wetlands. Steyer (1993) showed that terracing at Sabine National Wildlife Refuge increased primary productivity through the creation of emergent marsh (building of a terrace field) and subsequent expansion and colonization of emergents into adjacent open water areas.

Table 2: Wetland restoration and mitigation projects in coastal Louisiana where terraces were built or planned. Adapted from Nyman and Rohwer (unpublished), and Stead and Hill (2004).

Project Name	Marsh Type	Pond Acreage	Terrace Length (m)
LaBranche Wetlands Terracing (PO-28)	intermediate	489	21,330
Little Vermilion Bay Sediment Trapping (TE-12/PTV-19)	fresh	441	7,110
Plowed Terrace Demonstration Project (CS-25)	intermediate	unknown	6,450
Brown Lake Hydrologic Restoration (CS-09)	brackish	282	7,630
East Sabine Lake Hydrologic Restoration (CS-32)	brackish	393	Unknown
Four Mile Canal Terracing and Sediment Trapping (TV-18)	fresh	327	19,500
Pecan Island Terracing (ME-14)	brackish	442	60,890
Sabine Terraces (CS-ST)	brackish	110	unknown
Sediment Trapping at the Jaws (TV-15)	fresh	1,999	18,600
Sweet Lake/Willow Lake Hydrologic Restoration (CS-11b)	fresh	247	23360
Grand-White Lakes Landbridge Protection (ME -19)	Fresh	213	5960
Delta Management at Fort St. Philip (BS-11)	unknown	267	1000
Oyster Lake Terracing, Marsh Island Refuge	brackish	unknown	4,430
Cameron Creole NWR	brackish	unknown	unknown
Rockefeller State Wildlife Refuge	brackish	59	unknown
Sabine NWR, Unit 6	brackish	unknown	16,000
Sabine NWR, Unit 7	brackish	unknown	6,020
Sweet Lake	brackish	unknown	unknown
Falgout Canal Flotant	fresh	11	unknown
Delcambre Terrace Demo	unknown	12	1,645
Delcambre Terraces 2	unknown	7	unknown
Audubon Terraces	intermediate	10	unknown
DU Terrace Demo	unknown	107	14170
Smooth Cordgrass Maintenance Demonstration: Black Bayou	unknown	unknown	9370
DU Terrace Top Demo	unknown	25	unknown
DU Terraces -Hackberry	unknown	28	unknown
Apache Terrace Tops	unknown	18	unknown
	Total	5487	223,465

Additionally, three studies showed that terraced edge had more nekton biomass than did open water controls (Rozas and Minello 2001, Thom et al. 2004, Gossman 2005). Two of these studies (Rozas and Minello 2001, Thom et al. 2004) were unreplicated and specific to only Sabine NWR, and thus extrapolating results to the entire Chenier Plain may be inappropriate. Results from these studies additionally suggest that terracing changes nekton community composition. Gossman (2005) also suggested that body condition of nekton may be less at restored sites than unrestored sites, because organic matter also was less abundant at newly constructed terrace edges than at unrestored sites. Only one study has examined terrace effects at multiple sites and whole-pond scales (Cannaday 2006); he concluded that terracing increased submerged aquatic vegetation abundance at both the microhabitat and whole-pond scale. The efficacy of terraces at improving habitat quality for waterbirds, which depend heavily on coastal marshes, has never been evaluated.

I evaluated the quality of terraced ponds as waterbird habitat by comparing waterbird density and species richness at microhabitat and whole-pond scales in restored and unrestored ponds. I also evaluated whether bird density varied by foraging guild in restored and unrestored ponds. Additionally, to test assumptions of restoration managers, I compared SAV, nekton, and water quality variables at the microhabitat and whole-pond scales in restored and unrestored ponds. Finally, I evaluated which of those variables influenced waterbird density in terraced marsh.

METHODS

Study Area

My study sites were in coastal southwestern Louisiana within the Chenier Plain. This region of the Gulf Coast extends from Vermilion Bay, Louisiana, west to East Bay, Texas. It consists of shore-parallel, stranded inland beach ridges separated by broad areas of low elevation marsh. The Chenier Plain was formed by the long-shore westward transport of sediments from the Atchafalaya, Mississippi, and other rivers. These sediments were deposited in progradational mudflats along the shoreline. As the flats reached sufficient elevation, they were colonized by marsh vegetation. When the river delta shifted eastward, sediment was no longer deposited. Marine action gradually eroded the mudflats, and reworked the coarser grained sediments and mollusk shell, into higher elevation transgressive beach ridges, termed "cheniers." This process was cyclical, as the rivers shifted from east to west many times, creating a series of linear chenier ridges separated by broad areas of marsh (Penland and Suter 1989).

Marsh in coastal Louisiana generally is classified into four types based on characteristic dominant plant communities (Penfound and Hathaway 1938, Chabreck 1970). These types, in order of salinity, are saline, brackish, intermediate, and fresh. These marsh types generally occur in bands parallel to the shoreline, with the saltiest zones close to the gulf and the freshest zones being most interior (Gosselink 1979). As of 1998, the Chenier Plain consisted of 135 km^2 saline marsh (4% of total marsh area), 803 km^2 brackish marsh (26%), 684 km^2 intermediate marsh (22%), and 1435 km^2 fresh marsh (47%) (Louisiana Coastal Wetlands Conservation and Restoration Task Force and Wetlands Conservation and Restoration Authority 1998).

I monitored ponds dominated by *Spartina patens* at study sites located throughout the Chenier plain. At each site, I monitored one terraced pond (treatment), and one nearby unterraced pond (control). Each pair is at a different site, and hydrologically distinct from the others (Figure 1).

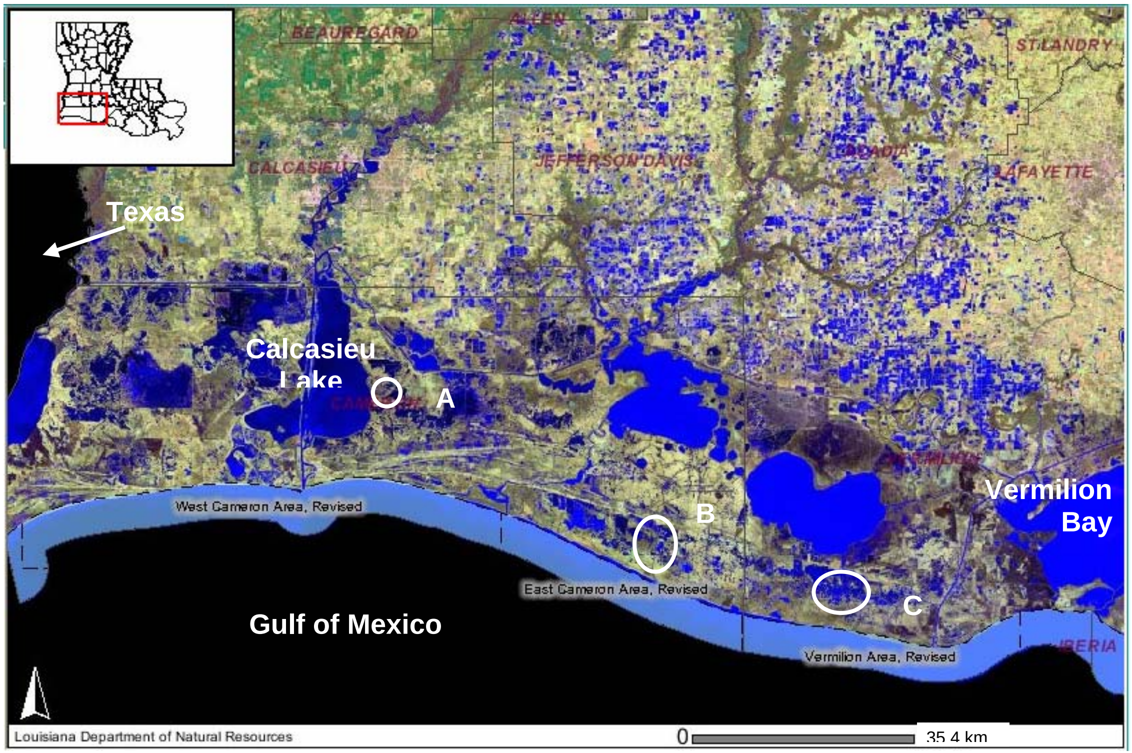


Figure 1. Louisiana's Chenier Plain, showing locations of sites used in surveys during spring and summer 2005. A: Sweet Lake site; B: Rockefeller SWR site; C: Vermilion site

Site Selection

Study ponds were selected by first identifying marsh dominated by *Spartina patens*, which is an indicator of intermediate and brackish marsh types. From these, I excluded sites receiving major mineral sediment sources. Most wetlands in the Chenier plain do not receive large amounts of sediments from rivers, streams or bayous. Additionally, sites were picked only if they were known to have been emergent marsh prior to 1956. This last criteria was based on land change maps (1956-1990) created by Barras et al. (1994). Additionally, the terraces within sites also had to be mature enough to have established emergent vegetation. Finally, sites were included only if terraced and unterraced ponds were close to each other, of similar size, salinities, and under similar hydrologic regimes (i.e. undergoing the same water management scheme, or otherwise having similar water inputs). This left me with three sites, which were the only appropriate available pond pairs left in the Chenier Plain. All of these were included in the study.

Site Monitoring Effort Complications

In the initial stages of my research, I sampled a second terraced pond within Vermilion Parish. It was subsequently decided that these ponds were not hydrologically distinct from the other ponds in Vermilion Parish. Data collected from the second Vermilion pond pair were analyzed as plot replicates within the Vermilion site (see statistical methods).

Site Descriptions

Sweet Lake- These study ponds were located in Cameron Parish, LA, near Grosse Savanne hunting lodge (1730 Big Pasteur Rd., Lake Charles, LA 70607). The land was owned by Sweet Lake Gas and Oil Co., Miami Corporation, and Grosse Savanne Waterfowl & Wildlife Lodge. Calcasieu Lake borders the site to the west, and Sweet Lake borders it to the east. The terraced pond was directly north of the unterraced one. They were separated from each other by two spoil banks, which have a canal running between them (Figure 2). Both ponds were equidistant from Calcasieu Lake, the major source of saline water in the area. They were thus under similar hydrologic regimes. The terraces were constructed in 2001.

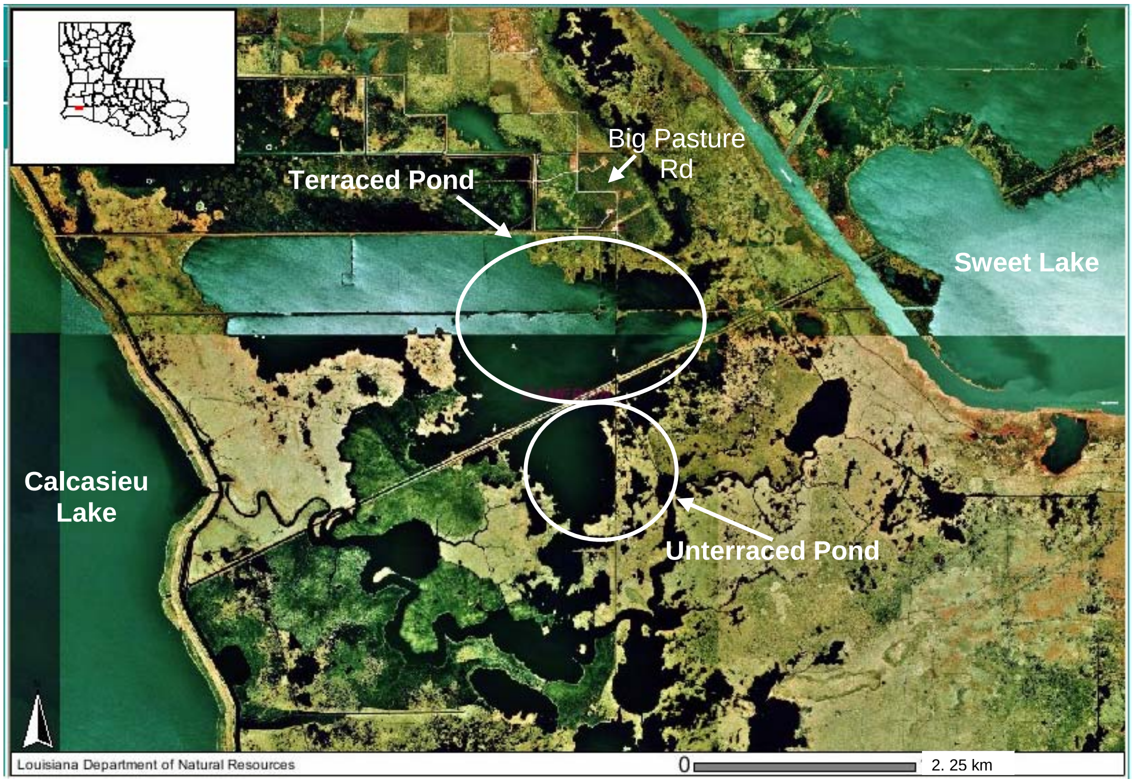


Figure 2. Sweet Lake survey ponds, Chenier Plain, Louisiana.

Rockefeller State Wildlife Refuge- The refuge is in southeastern Vermilion and southwestern Cameron Parishes (Hwy. 82, Grand Chenier, LA 70643). The land is owned by Louisiana Department of Wildlife and Fisheries. The refuge is situated between the Gulf of Mexico (to the south) and the Grand Chenier Ridge Complex, six miles inland (Melancon et al. 2000). The average elevation of the marsh in this area is 0.3 m above mean sea level (Chabreck 1960). The study ponds were located in Unit 4, an area of brackish impounded marsh actively managed for waterfowl. The terraced pond was directly north of the unterraced one (Figure 3). These terraces were constructed in 2002.

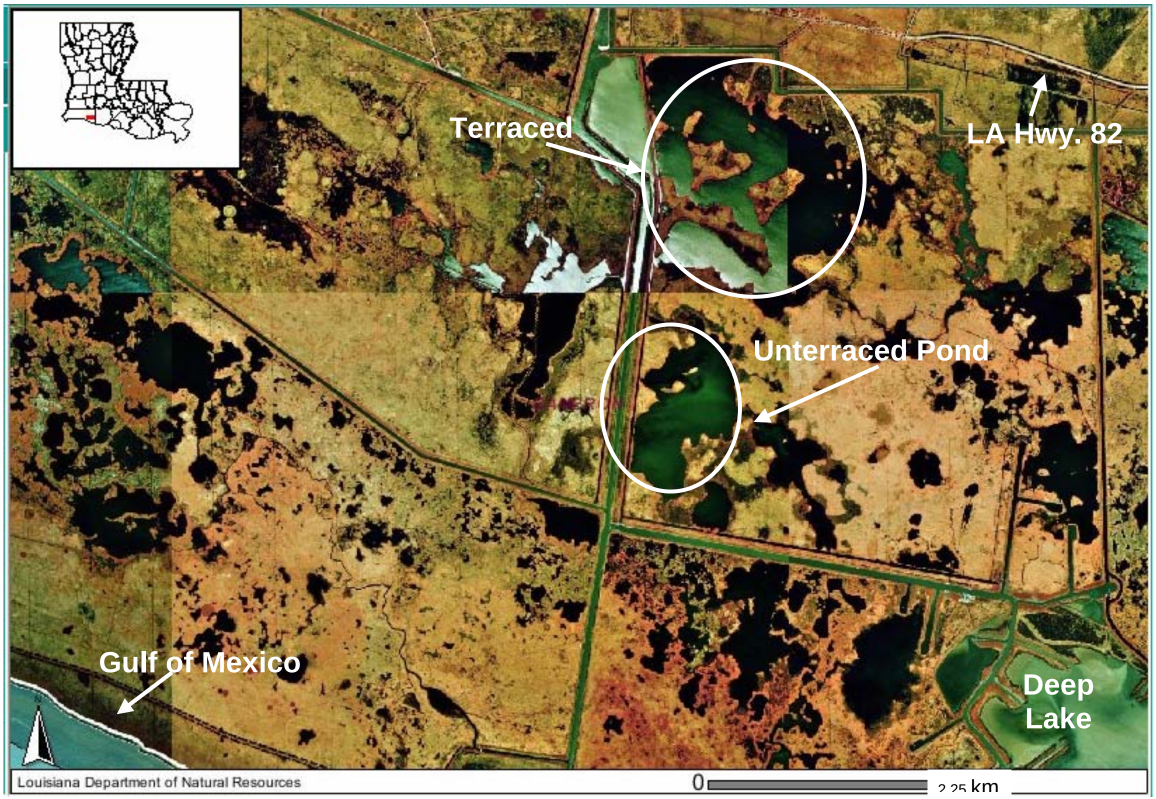


Figure 3. Rockefeller State Wildlife Refuge, unit 4, Chenier Plain, Louisiana.

Vermilion- Two pond pairs were in an area of marsh south of Pecan Island, LA, owned by Vermilion Corporation (115 Tivoli St., Abbeville, LA 70510), in Vermilion Parish. The area is bordered by LA Hwy 82 to the north, Rockefeller SWR to the west, and the Gulf of Mexico to the south (Figure 4). These terraces were constructed in 2003.

Survey Methods

Surveys were conducted from April 29, 2005 through September 3, 2005. Sampling frequency was once a month. Each pond contained multiple survey plots. Before surveys began, locations for plots were randomly selected, and boundaries were marked with pvc pipe.

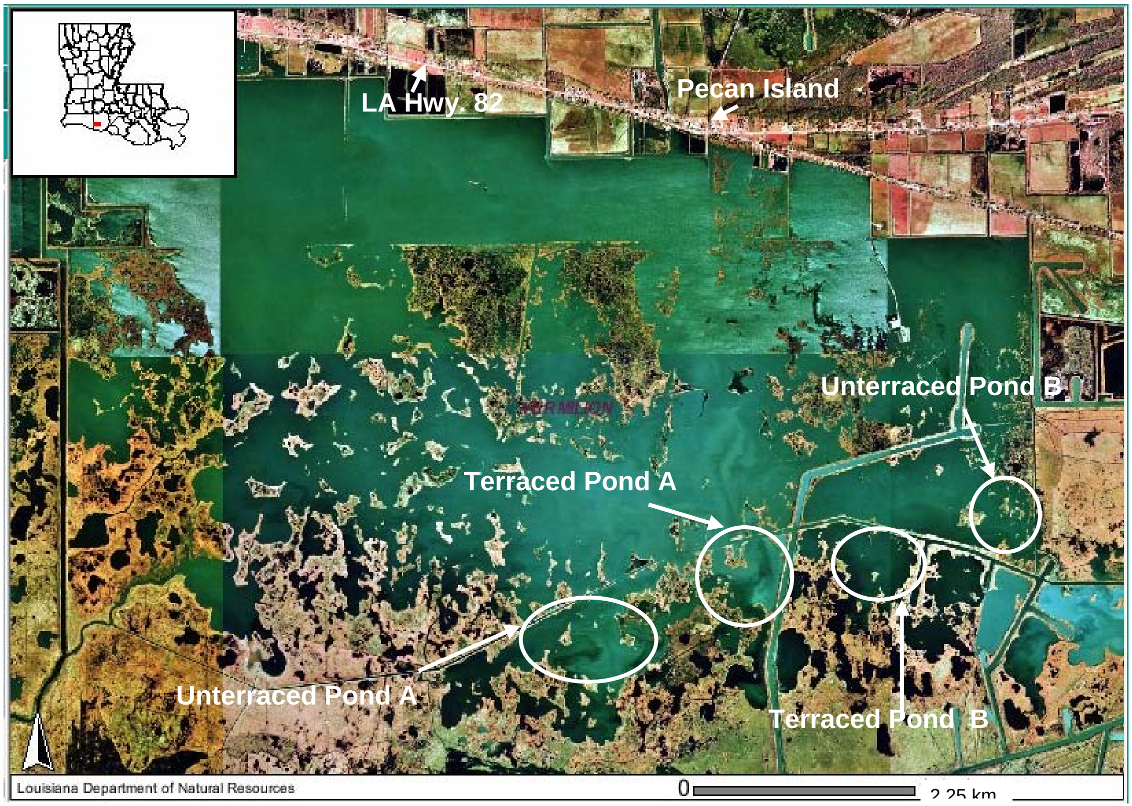


Figure 4 .Vermilion Parish survey ponds A and B, Chenier Plain, Louisiana.

Plots were originally designed to be of equal size (approximately 12 hectares), but at sites added later in the study, some plots were smaller because of geographical constraints. Thus, size of all plots ranged from 4 to 12 hectares. One plot per pond was sampled during each survey session. A different plot was sampled each survey session, such that sampling effort was even among plots. Each survey session contained water quality, nekton, and bird sampling.

Environmental Sampling

I measured wind speed, air temperature, water depth, salinity, conductivity, turbidity, and water temperature. One wind speed and air temperature measurement was recorded preceding each bird survey, using an EA-3010TWC anemometer (La Crosse Technology, 1116 South Oak Street, La Crescent, MN 55947 USA). The other environmental variables were measured following each bird survey in two microhabitat types: marsh edge, and open water greater than 25 m from any edge. As with nekton and SAV sampling, edge sampling was of terraced edge in the terraced pond, and of natural edge in the unterraced pond.

Two water depth measurements were taken within a 1-m^2 throw trap each time it was deployed (described below). This minimized variation in depth due to wave action. Salinity, conductivity, and water temperature were measured outside the throw trap, using an YSI model 63 (Yellow Springs Instruments Inc., 1725 Brannum Lane, Yellow Springs, OH 45387 USA). Turbidity was measured using an Oakton Instruments T100 Turbidity Meter Kit, model WD-35635-00 (Oakton Instruments P.O. Box 5136, Vernon Hills, IL USA 60061), calibrated prior to each use. Turbidity samples were collected far from SAV and nekton sampling locations, in undisturbed water.

Nekton and Submerged Aquatic Vegetation Sampling

Submerged aquatic vegetation and nekton were sampled in two microhabitat types: less than 5 m from emergent vegetation edge (the natural edge in unterraced ponds, and the terraced edge in terraced ponds), and greater than 25 m from the edge in open water. Specific sampling locations within the plot were chosen by taking a random compass direction, and then using the nearest sampling point of appropriate microhabitat in that direction. Submerged aquatic vegetation and nekton samples were collected using a 1-m^2 by 0.66-m high throw trap (Figure 5). This method is commonly used for sampling decapods, small adult fish, and juveniles of large fish (Kushlan 1981, Sogard and Able 1991, Raposa and Roman 2001). The trap was constructed of a welded aluminum frame similar to that described in Kushlan (1981). The trap is heavy and sinks rapidly to the bottom of the pond when thrown. The frame is covered by mesh cloth (mesh size= 1.6-mm). The mesh extends a further 0.25-m beyond the metal frame, and was supported by buoyant 1-m^2 PVC piping. This lengthened the height of the trap if it was thrown into water greater than 1 m. The trap was thrown from the bow of the boat. It was quickly pressed down into the sediment to prevent animals from escaping. A bar seine (size 1-m by 0.5-m, 1.6-mm mesh) was used to remove captured nekton from the trap. The bar seining was conducted from two sides of the trap until five consecutive passes yield no nekton. Nekton harvested in this fashion were put on ice, taken to the lab, counted, identified, and weighed. Additionally, all submerged aquatic vegetation within the trap was collected by hand. In the lab, the SAV was sorted by genus, dried, and weighed.

Figure 5. 1-m^2 throw trap, used for sampling small nekton and juveniles of large nekton species, being thrown from the bow of the boat next to the terrace edge at Rockefeller SWR, Chenier Plain, Louisiana.

Bird Surveys

Surveys began at dawn and continued for 90 minutes. Observers arrived via boat and allowed a 15-minute settling period before beginning observations. During surveys, observers sat hidden in emergent vegetation, using camouflage netting for additional cover. Two observers were used so that simultaneous observations of terraced and unterraced ponds were possible. Observers rotated equally between pond types on subsequent survey sessions, to spread any variation resulting from observer bias between treatments.

Observers recorded bird abundance and diversity every 15 minutes, generating seven bird counts per survey. Additionally microhabitat used (emergent vegetation, mudflat, natural edge, terraced edge, or open water) and qualitative distance to nearest cover (near: within 5 m of cover, intermediate: within 15 m of cover, or far: greater than15 m from cover) were recorded. For flocks of more than ten individuals per species, total flock size per species was recorded, but behavior and microhabitat details were taken for only a subset of ten individuals. Only data for birds actively using the pond were recorded. Data for aerial foragers flying over the pond were recorded only after foraging behavior was exhibited (diving on the pond and subsequently circling over it).

Statistical Methods

All statistical analyses were conducted using SAS 9.1.2 (SAS Institute Inc., 100 SAS Campus Drive, Cary, NC 27513-2414, USA). Multiple days were required to sample all sites. However, for the purpose of analysis, I assigned each survey a single date (the average of the dates over which the survey took place). For analysis, I classified the second Vermilion Parish pond pair as plots within the Vermilion Parish site because it was not hydrologically distinct from the other ponds sampled in Vermilion Parish. Thus, on surveys where both pairs of Vermilion ponds were sampled, they were analyzed as day-site replicates of each other.

Pond Characterization

To compare the proportion of different microhabitat types available in each pond type, I analyzed 2004 DOQQ aerial photographs (Figure 6 a) of all sites using ArcGIS 9.1 (ESRI Corporation, 380 New York Street, Redlands, CA 92373-8100, USA). I classified each portion of the pond as either edge or open water habitat. To do this, I defined a pond as consisting of only water. Any areas of emergent vegetation, whether natural or on terraces, were excluded from pond area. I then classified any portion of this watery pond within 10 m of an emergent vegetation edge as edge habitat, and all the rest of the water in the pond as open water habitat (Figure 6 b). I used ArcMap to generate the area of each microhabitat type, and then converted this into a percent of total pond area. Percent edge was not normally distributed. I used the nonparametric Wilcoxon Rank Sum test to compare proportion of available edge habitat between pond types. I used a logistic regression with pond type as the response variable and water quality variables as dependant factors to compare water quality between pond types.

Nekton and SAV Analyses

I used a repeated measures ANOVA with blocking on site to compare nekton density and SAV biomass between pond types (terraced or unterraced). I additionally included the microhabitat (open or edge) in which the samples were collected as an independent factor in the model (Table 3).

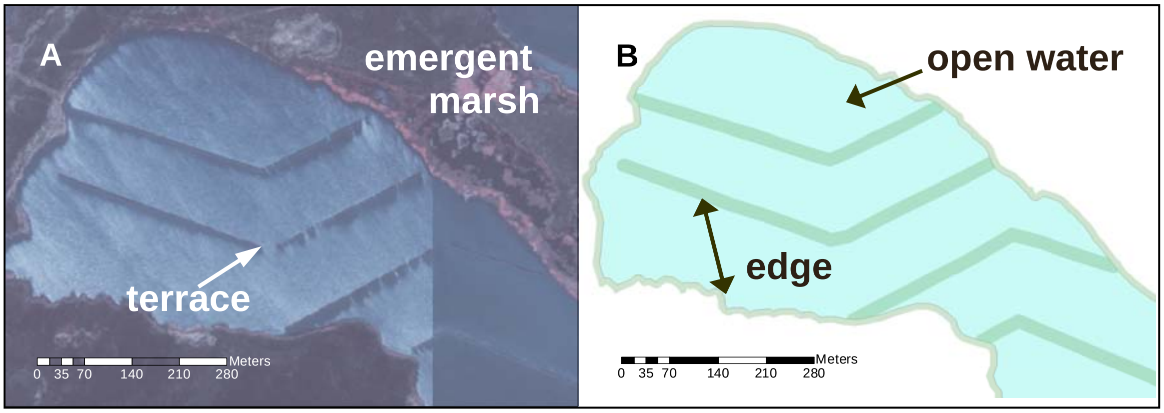


Figure 6. 2004 DOQQ of Rockefeller SWR's terraced pond, Chenier Plain Louisiana (A) and ArcMap (ESRI Corporation, Redland, CA) microhabitat classification (B) of that pond into open water area and emergent marsh edge area. Terraced edge and natural edge were combined into one edge class.

Two *a priori* contrasts were analyzed, comparing nekton density at marsh edge in terraced and unterraced ponds, and comparing nekton density in open water in terraced and unterraced ponds. The exact same *a priori* contrasts were used to compare SAV biomass in these treatments. Residuals were examined and response variables were log transformed to achieve normality and reduce heterogeneity of variances.

I performed a backwards stepwise regression to determine whether any measured variables could explain variation in nekton density without including pond type in the model. Potential explanatory variables were day, SAV biomass, water temperature, turbidity, conductivity, water depth, and proportion of microhabitat types (edge or open) in ponds. Average nekton and SAV for the whole pond were used for this analysis. This was determined by multiplying near and far sample means by the proportion of pond that was edge or open habitat, respectively. Residuals were examined and results were log transformed to achieve normality of residuals and improve heterogeneity of variances.

Table 3. Experimental design of study to determine the effects of terraces on nekton/m^2 or SAV g/m^2: an ANOVA with blocking on site and repeated measures. All site interactions were pooled into the error term *a priori*. Design is not balanced. On three surveys, one inaccessible site was not sampled due to low water conditions. Thus, actual degrees of freedom differ from those listed.

Factor	N	*df*	levels
Treatment (pond type)	2	1	Terraced, Unterraced
Date	6	5	
Microhabitat	2	1	Edge, Open
Date*Treatment		5	
Microhabitat*Treatment		1	
Microhabitat*Date		5	
Micro*Date*Treat		5	
Site	3	2	Sweet Lake, Rockefeller, Vermilion
Total		71	

Bird Analyses

To avoid double counting individuals, I used the greatest number of birds of a given species seen during any one count interval as my estimate of bird abundance for that species for each survey. I converted bird abundance to bird density by dividing bird abundance by plot area. Species richness was defined as the number of species observed during a survey. I used a repeated measures ANOVA with blocking on site to compare bird density and species richness between terraced (treatment) and unterraced (control) ponds (Table 4). Residuals were examined and I log transformed bird density to achieve normality and reduce heterogeneity of variances.

I classified bird species into guilds based on foraging method to evaluate if birds density in pond types varied among foraging guilds. To classify birds, I generally used the foraging classifications proposed by De Graaf et al (1985). My classification scheme differs somewhat from that of De Graaf et al (1985). I categorized American White Pelican as divers, although they never dive. I categorized them as divers because the ponds are shallow and their long necks enable them to forage lower in the water column than other surface foragers. De Graaf et al. (1985) describes Common Moorhens as both divers and dabblers, but I only observed them dabbling in our ponds, so I categorized them exclusively as dabblers. The resulting guilds are as follows:

1. Diving foragers: grebes, diving ducks, cormorants, American White Pelican
2. Wading foragers: herons, egrets, ibis, and Roseate Spoonbill

3. Shorebirds and other probers/surface arthropod gleaners: sandpipers, plovers, American Avocet, Black-Necked Stilt, and rails4. Aerial foragers: terns, gulls, and Belted Kingfisher

5. Dabblers: dabbling ducks, Common Moorhen, American Coots, and Purple Gallinule

I compared guild density between pond types, using a repeated measures ANOVA with blocking on site (Table 4). I examined the residuals of the results. For most guilds, log transformations were necessary to obtain normal response variables and improve homogeneity of variances. It was not necessary to log transform wading foragers. When significant pond type by time interactions were seen, six post hoc tests with tukey adjustments were used to compare responses between pond types on each survey date.

A number of species of conservation concern are known to occur in Louisiana's coastal brackish marshes. It is possible that pond terracing effects habitat for species of concern differently than it effects habitat for other species. For this analysis, I used the conservation classifications proposed by the Waterbird Conservation Council (Kushlan et al. 2002). These include 13 Gulf Coast species that are classified as species of high concern (Black Skimmer, Least Tern, Little Blue Heron, Snowy Egret, Tricolored Heron, Gull-billed Tern, Roseate Tern, Pied-billed Grebe, Purple Gallinule, American Bittern, King Rail). It additionally includes 14 Gulf Coast species classified as species of moderate concern (American White Pelican, Forster's Tern, Anhinga, Neotropic Cormorant, Reddish Egret, Roseate Spoonbill, White Ibis, Black-crowned Night-Heron, Eared Grebe, Royal Tern, Clapper Rail, Virginia Rail, Common Moorhen, Common Loon).

Table 4. Experimental design of study to determine the effects of terraces on waterbird density and richness: an ANOVA with blocking on site and repeated measures. All site interactions were pooled into the error term *a priori*. Actual degrees of freedom differed from anticipated. One site contains two terraced/unterraced pond pairs instead of one. This is because it was decided after sampling had begun that they were not hydrologically distinct from each other and represented one site rather than two. Once the airboat broke and Rockefeller was not sampled (site is impounded and an airboat is required).

Factor	N	*df*	levels
Treatment (pond type)	2	1	Terraced, Unterraced
Date	6	5	
Date*Treatment		5	
Site	3	2	Sweet Lake, Rockefeller, Vermilion
Total		35	

To be included in analysis, a species had to have been observed on at least three separate occasions. I compared density of each of these species of concern between pond type, using a repeated measures ANOVA with blocking on site (Table 4). Residuals of results were examined, and log transformations were necessary to improve normality of response variables and homogeneity of variances. When significant pond type by time interactions were seen, six post hoc tests with tukey adjustments were used to compare responses between pond types on each survey date.

Additionally, I used a backwards stepwise regression to determine whether any measured variables could explain variation in bird density of all birds without including pond type in the model. Potential explanatory variables were nekton density, SAV biomass, water temperature, turbidity, salinity, conductivity, water depth, air temperature, wind speed, and proportion of microhabitat types (edge or open) in ponds.

Finally, I analyzed whether individual birds were observed more often in edge or open water microhabitats. To ensure independence, I only used one bird count per survey. I preferred to use bird counts from the middle of surveys. Data from the middle of surveys was probably most representative of natural behavior because birds were most acclimated to the disturbance of our arrival, and because it was still early enough that bird activity levels were high. I used, in order of preference, data from count 4, 5, 3, 6, 7, 2, or 1, until I found an interval in which birds were observed. If an individual bird was within 5 m of emergent vegetation on any side, I classified it as using edge habitat. Otherwise, I classified it as using open water habitat. I then compared the number of birds in edge versus open microhabitats using a chi-square test.

RESULTS

Pond Characterization

None of the water quality variables measured (turbidity, water depth, salinity, conductivity, water temperature) differed between terraced and unterraced ponds regardless of microhabitat (Table 1). Ponds averaged 80 hectares (range 9 to 470 hectares). Adding terraces to ponds increased the proportion of pond edge habitat (Table 6). Terraced ponds had significantly more edge habitat (40%, se = 3%) than did unterraced ponds (8%, se = 1%) when terraced edge and natural edge were combined into one marsh edge category ($p < 0.0001$).

Nekton and SAV

Whole-Pond Analysis

Adding terraces to ponds did not significantly increase nekton density/m^2 or SAV biomass (g/m^2)) at the whole-pond scale ($F_{1,\,14} = 0.21$ $p = 0.66$ and $F_{1,\,14} = 0.41$, $p = 0.53$, respectively). Mean log of nekton/m^2 was 2.6, se = 0.48, (raw average = 53.9, se = 25.6) in terraced ponds, and was 2.8, se = 0.47, (raw average = 44.8, se = 15.3) in unterraced ponds. Mean log of SAV biomass (g/m^2) was 1.34, se = 0.49, (raw average = 14.4, se = 5.7) in terraced ponds, and was 1.49, se = 0.49, (raw average = 12.6, se = 4.2) in unterraced ponds.

Nekton density was significantly influenced by date, SAV biomass and conductivity in ponds ($F_{3,23} = 49.3$, $p<0.0001$). The regression equation which best explained variation in nekton density was: log (nekton/m^2) = 0.59 * log (SAV g/m^2) - 0.045 * conductivity + 0.016 * day - 0.41 ($R^2 = 0.84$).

Table 5. Mean water quality (+/- se) for marsh edge and open water habitats in terraced and unterraced ponds, spring and summer of 2005, Chenier Plain, Louisiana. Water quality did not differ significantly between pond types.

Variable	Microhabitat	Terraced Pond	Unterraced Pond
Turbidity (NTU)	open water	26.4 +/- 4.5	52.6 +/- 19.7
	marsh edge	55.8 +/- 20.6	41.6 +/- 7.5
Water depth (cm)	open water	40.3 +/- 4.7	44.4 +/- 3.9
	marsh edge	27.2 +/- 3.8	26.7 +/- 3
Salinity (ppt)	open water	6.33 +/- 1.27	7.82 +/- 1.65
	marsh edge	6.5+/- 1.2	8.2 +/- 1.7
Conductivity (mS)	open water	11.6 +/- 2.3	14.7 +/- 2.7
	marsh edge	11.8 +/- 2	13 +/- 2.4
Water temperature (ºC)	open water	26.2 +/- 1.4	25.7 +/- 1.3
	marsh edge	26.2 +/- 1.5	25.9 +/- 1.2

Table 6. Area (hectares) of pond within 10 m of a vegetated edge for both edge types (terraced and natural) in terraced ponds at all sites.

Site	Terraced Edge	Natural Edge	Total Edge	Terrace to Natural Edge Ratio
Vermilion	19.1	2.5	21.6	7.5
Little Vermilion	1.7	1.7	3.4	1.0
Sweet Lake	50.5	10.3	60.8	4.9
Rockefeller	7.3	5.1	12.3	1.4

Microhabitat Analysis

Terraced ponds and unterraced ponds had similar nekton density and SAV biomass at the marsh edge (Figure 7a and b, $F_{1, 32} = 0.63$, $p = 0.43$ and $F_{1, 29} = 0.01$, $p = 0.94$ respectively). Similarly, the two pond types have similar nekton density and SAV biomass in open water (Figure 7a and b, $F_{1, 32} = 0$, $p = 0.95$ and $F_{1, 29}=0.41$, $p=0.53$ respectively). When data from terraced and unterraced ponds are combined, nekton density ($F_{1, 29} = 15.21$, $p = 0.0005$) and SAV biomass ($F_{1, 39} = 5.7$, $p = 0.022$) were significantly different between microhabitat types. Mean log of nekton/m^2 was 3.12, se = 0.6, (raw average = 44.45, se = 12.4) in open water habitat, but was 2.1, se = 0.6, (raw average = 104.1, se = 11.8) in marsh edge habitats. Likewise, mean log of SAV biomass (g/m^2) was 1.4, se = 0.5, (raw average = 11.3, se = 3.1) in open water habitat, but was 2.0, se = 0.5, (raw average = 20.8, se = 3) in marsh edge habitats.

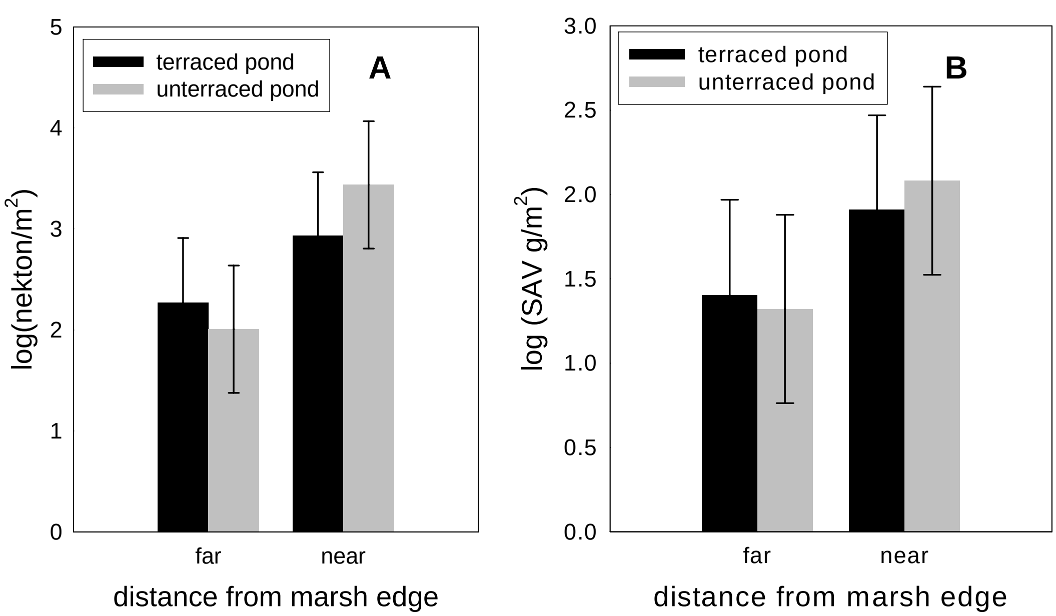


Figure 7. Nekton density (A) and SAV biomass (B) in terraced and unterraced ponds at two microhabitat types, in spring and summer of 2005, Chenier Plain, Louisiana.

Bird Results

Whole-Pond Analysis

Figure 8 presents the raw data for waterbirds, summed for all surveys. To give a visual representation of species richness, each wedge represents relative frequency for bird species that were seen more than twice. Bird species seen less often are lumped together into "other." The area of the pies is proportional to total bird density (birds/hectare) from all surveys for the two pond types.

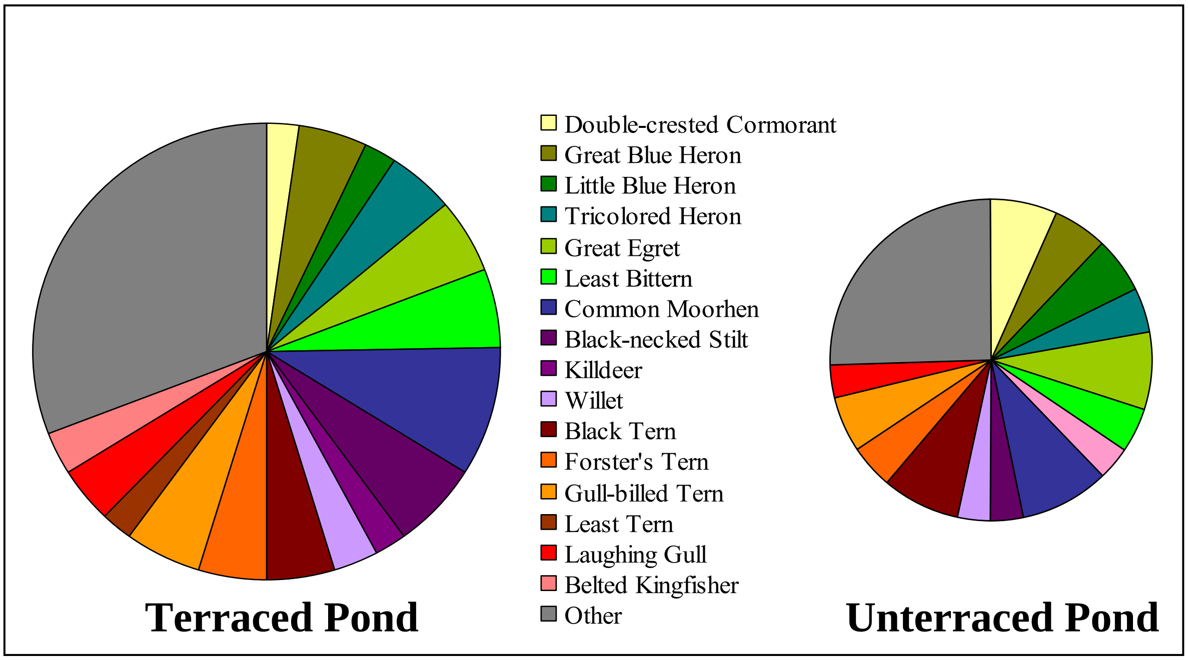


Figure 8. Relative frequency and species richness for bird species in terraced and unterraced ponds summed over all surveys, in spring and summer of 2005, Chenier Plain, Louisiana. Size of charts is proportionate to total density (birds/hectare) for that pond type. Birds seen less than twice are lumped into "other."

Average raw bird density in terraced ponds was 3.4 birds/hectare (se = 0.9) and was 0.9 birds/hectare (se = 0.2) in unterraced ponds, a 75 % difference. For reference, raw bird density through time is depicted in Figure 9.

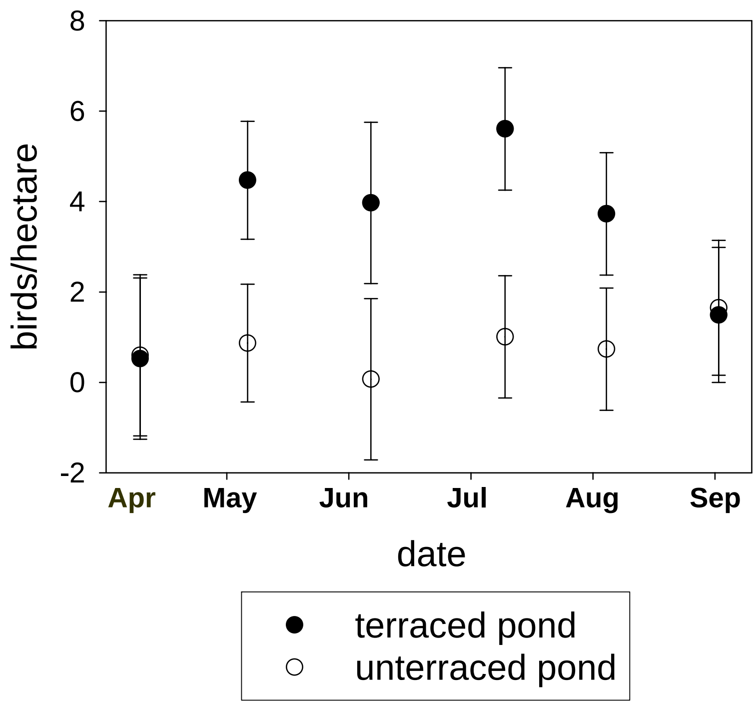


Figure 9. Raw bird density (mean +/- se) in terraced and unterraced ponds, in spring and summer of 2005, Chenier Plain, Louisiana. These data are for reference and were not the statistical model analyzed.

Bird density was greater in terraced ponds (Figure 12, $F_{1, 22}$ =22.95, $p < 0.0001$). Mean log of bird density was 1.2 birds/hectare (se = 0.13) in terraced ponds, and was 0.5 birds/hectare (se = 0.13) in unterraced ponds. Bird species richness differed between pond types at most times ($F_{5, 22}$ = 12.09, p=0.0021). Terraced ponds usually had greater species richness than unterraced ponds (Figure 10). On one survey in early September, unterraced ponds had greater species richness.

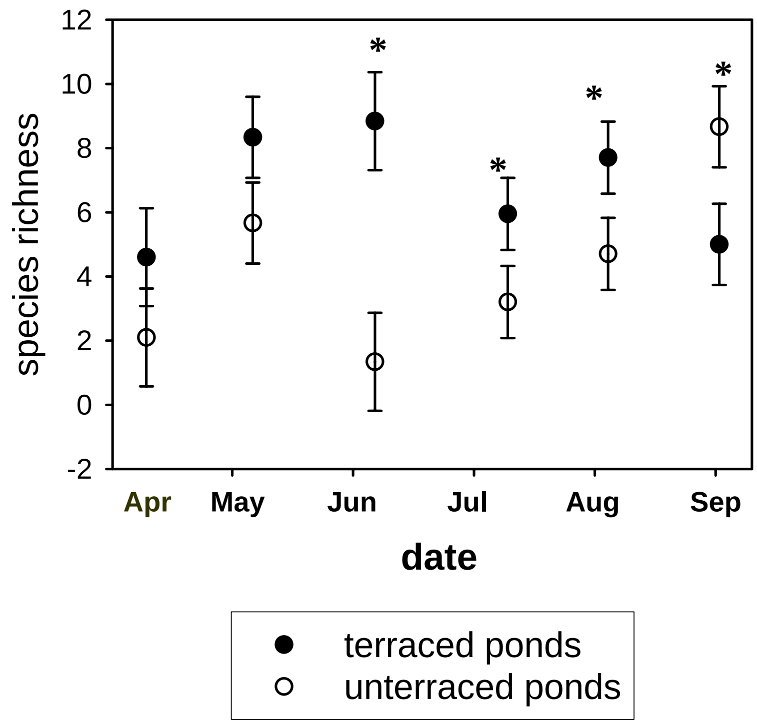


Figure 10. Bird species richness through time in terraced and unterraced ponds (mean +/- se), in spring and summer of 2005, Chenier Plain, Louisiana. Statistically significant differences are marked with an asterisk.

Guild Response

Figure 11 presents bird density and relative frequency by guild. A visual examination of this raw data suggests that relative frequency of bird guilds is similar between pond types, but that birds generally are more abundant in terraced ponds than unterraced. When pond types are combined, guilds were, in order of highest average density (birds/hectare) to lowest: shorebirds (0.6, se = 0.3), aerialists (0.5, se= 0.1), waders (0.3, se = 0.05), dabblers (0.3, se = 0.07), and divers (0.07, se = 0.02).

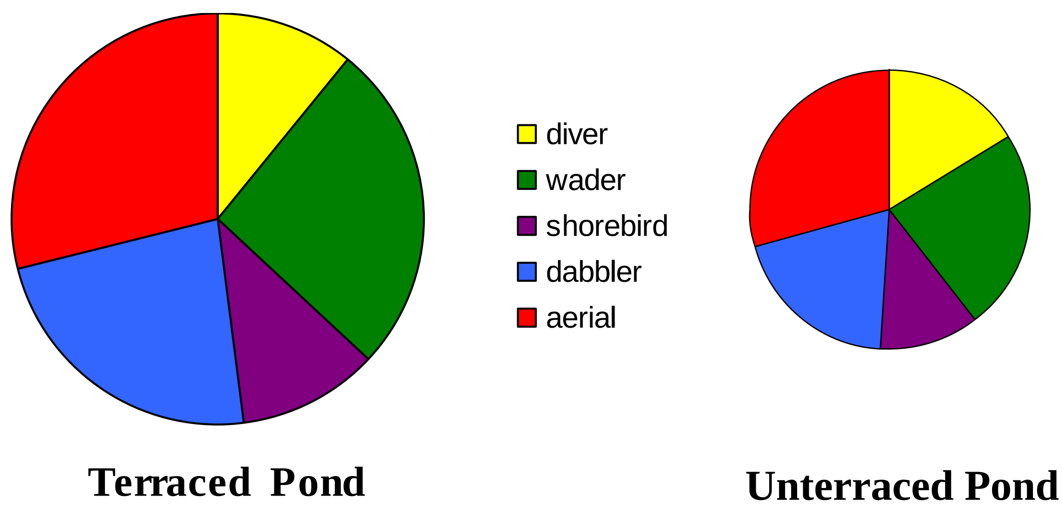


Figure 11. Raw density and relative frequency of foraging guilds in terraced and unterraced ponds, in spring and summer of 2005, Chenier Plain, Louisiana. The area of the pies is proportionate to total bird density (birds/hectare) from all surveys.

Birds of different foraging guilds differed in response to pond type. Three guilds (shorebirds, aerialists, and dabblers) had greater densities (birds/hectare) in terraced ponds than in unterraced ponds (Figure 12, $F_{1, 22} = 7.53$ $p = 0.01$, $F_{1, 22} = 7.14$ $p = 0.01$, and $F_{1, 22} = 4.55$ $p = 0.04$ respectively). Raw average density for shorebirds was 1.2, se 0.6, in terraced ponds, and 0.05, se 0.02, in unterraced ponds. Aerialist raw average density was 0.72, se 0.18, in terraced ponds and was 0.33, se 0.15, in unterraced. Dabbler raw average density was 0.4, se 0.12, in terraced ponds and was 0.18, se 0.07, in unterraced. For wading foragers, there was a significant pond type by time interaction (Figure 13, $F_{5, 22} = 7.02$, $p = 0.0005$), though they tended to be more abundant in terraced ponds. Raw mean density for wading foragers was 0.48, se 0.09, in terraced ponds, and was 0.2, se 0.05, in unterraced ponds. Log transformation was not necessary for wading forager analysis. Diving foragers were not abundant and density did not vary between pond types. Raw mean diver density was 0.08, se 0.03, in terraced ponds, and 0.07, se 0.03, in unterraced ponds.

Species of Concern Response

Seventeen species of conservation concern were observed in my study ponds (Table 7). Many of these species were not dense and were observed only rarely. Log transformations were necessary for all species to improve normality of model residuals. In most cases, normality of residuals was not achieved. Raw means for all species were either greater in terraced ponds or equal between pond types. Four species were significantly denser in terraced ponds (Common Moorhen, Gull-billed Tern, Tricolored Heron, and Least Tern), though only two of these had normally distributed model residuals (Gull-billed Tern, and Tricolored Heron). Three species had significant pond type by time interactions (Forster's Tern, Pie-billed Grebe, and Roseate Spoonbill). Forster's Terns were denser in terraced ponds on two occasions (Figure 14 a). Pie-billed Grebe and Roseate Spoonbill were denser in terraced ponds on one occasion and denser in unterraced ponds on another occasion (Figure 14 b and c, respectively).

Microhabitat Analysis

Birds preferred edge habitat. Edge habitat accounted for 26% (se = 0.031) of total available habitat, and 74% of birds were observed in edge habitat rather than open water ($\chi^2 = 7.3329$, df = 1, p = 0.0068). Finally, bird densities did not vary with any of the measured water quality variables (air temperature, or wind, submerged aquatic vegetation biomass, or nekton density), but did vary with proportion of edge habitat in ponds. Ponds with more edge had higher bird densities (Figure 15, $F_{1, 34} = 6.17$, p = 0.0181).

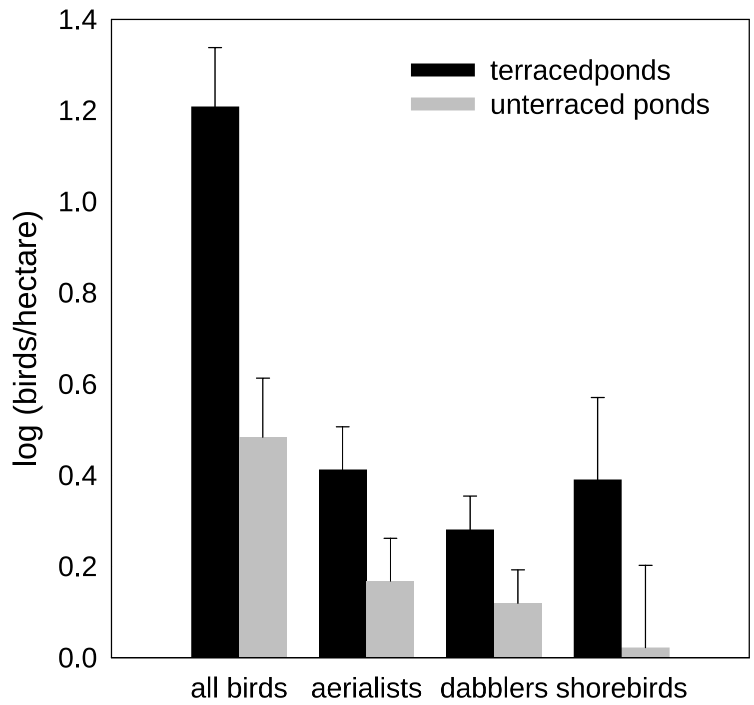


Figure 12. Log of density for all birds, aerialists, dabblers, and shorebirds in terraced and unterraced ponds (mean +/- se), in spring and summer of 2005, Chenier Plain, Louisiana.

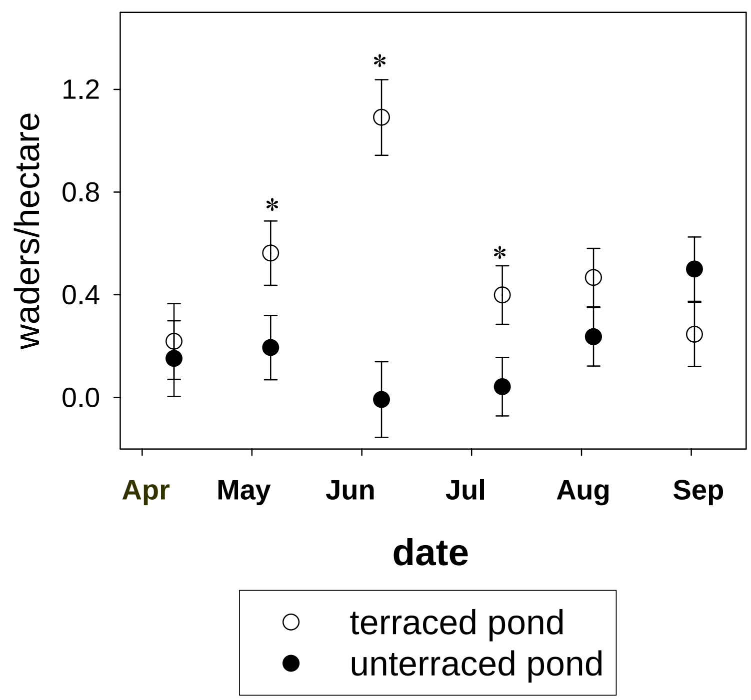


Figure 13. Wader density through time in terraced and unterraced ponds (mean +/- se), in spring and summer of 2005, Chenier Plain, Louisiana. Statistically significant differences are marked with an asterisk.

Table 7. Results for species of conservation concern, with raw means of density (birds/hectare) for pond type and standard errors. No. is the total number of separate occasions this species was seen. Sig. Effect is whether significant results were for pond type, pond by time interactions, or no significant effects were seen. F-statistic and p-value are for the significant effect listed. Normal is whether normality was achieved for model residuals after the response variable was log transformed.

Bird	Terraced	SE	Unterraced	SE	No.	Sig. Effect	F- statistic	p- value	Normal
Common Moorhen	0.39	0.13	0.09	0.03	20	Pond	$F_{1, 22} = 7.357$	0.012	no
Forster's Tern	0.14	0.06	0.02	0.01	10	Pond by time	$F_{5, 22} = 7.12$	0.0004	no
Gull-billed Tern	0.11	0.04	0.05	0.02	12	Pond	$F_{1, 22} = 3.94$	0.060	yes
Tricolored Heron	0.08	0.03	0.02	0.01	10	Pond	$F_{1, 22} = 6.52$	0.018	yes
Little Blue Heron	0.05	0.03	0.04	0.02	8	none			no
American Bittern	0.03	0.02	0	0	2	none			no
Least Tern	0.03	0.02	0	0	4	Pond	$F_{1, 22} = 5.7$	0.026	no
Snowy Egret	0.02	0.02	0.01	0.01	4	none			no
Pie-billed Grebe	0.02	0.01	0.02	0.01	4	Pond by time	$F_{5, 22} = 2.72$	0.046	no
Anhinga	0.02	0.02	0	0	2	none			no
King Rail	0.02	0.02	0.01	0.01	2	none			no
Roseate Spoonbill	0.01	0.01	0.01	0.01	3	Pond by time	$F_{5, 22} = 4.33$	0.007	no
Royal Tern	0.01	0.01	0	0	1	none			no
Clapper Rail	0.01	0.01	0	0	2	none			no
Purple Gallinule	0	0	0.01	0.01	2	none			no
Neotropic Cormorant	0	0	0	0	1	none			no
Reddish Egret	0	0	0.01	0.01	1	none			no

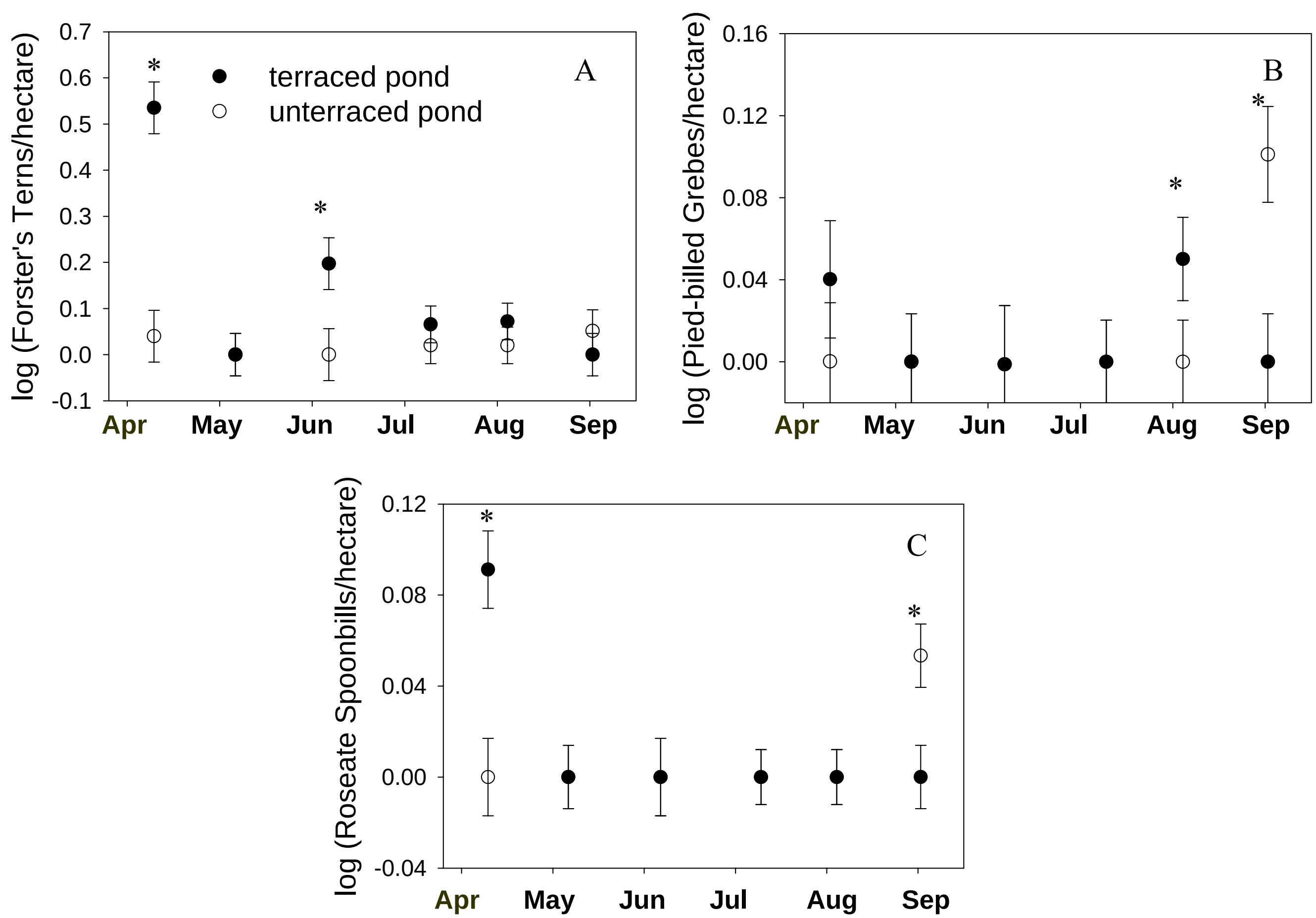
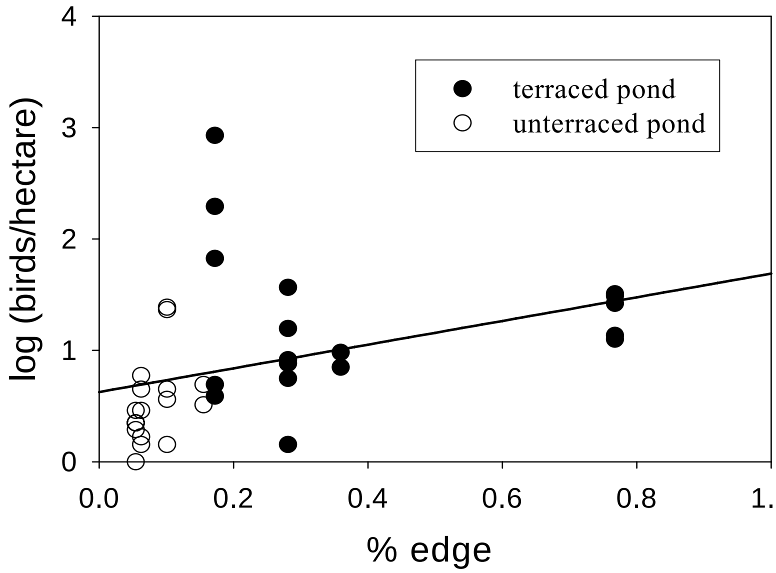


Figure 14. Forster's Tern (A), Pied-billed Grebe (B), Roseate Spoonbill (C) density through time in terraced and unterraced ponds (mean +/- se), in spring and summer of 2005, Chenier Plain, LA. Asterisks are for statistically significant differences.



$$\log(\text{birds/hectare}) = 1.06\ (\%\ \text{edge}) + 0.63,\ R^2 = 0.153$$

Figure 15. Relationship between log of mean bird density and percent available marsh edge habitat in ponds, in spring and summer of 2005, Chenier Plain, Louisiana.

DISCUSSION

Water quality (water temperature, salinity, conductivity, water depth and turbidity) did not differ significantly between pond types. This confirms assumptions that control ponds chosen were similar to terraced ponds in hydrology. Further, contrary to predictions about terrace effects, turbidity was not altered from unrestored conditions. Reduction of water turbidity often is cited as the mechanism through which submerged aquatics will be promoted by pond terracing. However, a more frequent sampling regime might have revealed a difference in water quality. Another prediction of terrace restoration is that terracing ponds should decrease water depth by encouraging sediment deposition, and thus dampen wave energy, slow erosion, and halt or slow marsh submergence. However, in this study, water depth was shown to be similar between pond types. This does not necessarily mean that terraces don't slow water movement. Unlike ponds studied by Steyer (1993) and Rosas and Minello (2001), ponds surveyed in this study were deliberately chosen to have no major outside sediment sources. This is a condition typical of most wetlands in the Chenier Plain. Thus, lack of significant sediment deposition is not very surprising. Additionally, all terraces studied were less than five years old. Terracing is a novel restoration tool, and nearly all terracing projects are less than five years old (Stead and Hill 2004). It is possible that given time and adequate opportunities from hurricanes and storm fronts bringing in offshore, or out-of basin, sediments that terracing would encourage greater sediment deposition than that seen in unrestored ponds. Some casual observations I took following Hurricane Rita suggest that this might be the case, but this has not been substantiated by this study or any other study.

However, peat production from emergent plants has been noted as the most likely mechanism to produce and maintain elevation changes in the Chenier Plain (Foret 2001), and peat production may be very important in maintaining marshes regionally (Nyman et al. 1993, Turner et al. 2000). Terrace restoration is predicted to encourage emergent vegetation expansion, yet interestingly, if and how, terraces affect emergent vegetation has been understudied. Steyer (1993) suggested that there was some lateral expansion of emergents into open water areas in terraced ponds at Sabine NWR, but this has not been examined by any subsequent study. Although not measured directly, there was no visually obvious expansion of emergents into open water adjacent to terrace edges in any of my study sites. Lateral expansion of emergents is necessary to reverse open water conversion of emergent marsh. Additionally, to prevent future open water conversions, vertical accretion of emergent marsh through *in situ* peat production may be necessary. Steyer (1993) suggested that there was mineral sediment accretion in one of the two terraced ponds studied at Sabine NWR, but these ponds were both close to a mineral sediment source, a rare condition in the Chenier Plain. If, and how, terracing encourages emergent peat production has never been thoroughly studied. Without the encouragement of emergent production laterally and vertically, over the long-term, any terrace effects are likely to be transitory.

Terracing increased the proportion of marsh edge habitat in ponds, with many resulting increases in the density of wildlife and submerged aquatics. Nekton and SAV were more abundant directly adjacent to terrace edges, and more over, there were no differences between natural marsh edge and terraced edge. Nekton densities were greater in areas where SAV was more abundant. Additionally, marsh edge contained greater nekton density and SAV biomass than did open water habitats. This supports results seen by Castellanos and Rozas (2001) who noted that nekton in an Achafalaya delta marsh were 92% denser when associated with marsh vegetation than when associated with bare surfaces. Additionally Minello and Rozas (2002) noted that in a Texas *Spartina alterniflora* marsh nekton were 56% denser 1 m from the marsh edge than they were 10 m away from the edge into the pond. However in my study, when averaged over the whole pond, terraced ponds did not contain significantly more nekton or SAV than did unterraced ponds. This may be because too few sites were sampled. Still, this is an interesting result, because terraced ponds contain more marsh edge (if terraced edge and natural edge are combined) than unterraced ponds do. I hypothesize that the lack of a pond level effect was likely caused by too few terraces built in all ponds sampled. Natural edge in my study ponds averaged 9% of pond area. Terraced edge averaged 25% of pond area, but ranged from 6 to 68%. The emergent marsh conditions historically seen in these ponds likely contained few areas of open water. Altering terrace designs to further increase the proportion of marsh edge and the area of created emergent marsh may more nearly mimic historic conditions. This may also produce pond level results for nekton and SAV.

Unlike nekton and SAV, birds responded to pond terracing at both the microhabitat and whole-pond scales. Bird densities were highest in edge habitat, and when averaged over the whole pond, density and species richness were greatest in terraced ponds.

Foraging guilds varied in their response to pond terracing. Shorebirds, aerialists, and dabblers, were consistently denser in terraced ponds than unterraced ponds, and wading foragers generally were denser in terraced ponds. Dabbling foragers likely benefited from pond terracing because of increases in SAV directly adjacent to terrace edge. Dabblers forage for SAV, SAV seeds, and aquatic invertebrates among SAV (Ehrlich et al. 1988). SAV also provides structure, refuge, and forage for nekton (Rozas and Odum 1988, Castellanos and Rozas 2001), and thus may provide a highly profitable foraging area for omnivorous and piscivorous birds as well. Shorebirds probably respond to pond terracing because they prefer to use edge habitat where water depths are generally shallower. Diving foragers did not seem to respond to pond terracing. However, all of these ponds were shallow (usually less than 1 m deep), and although diving foragers did utilize them, shallow ponds probably don't constitute important habitat for them.

It is important to recognize that while abundance of some forage items was analyzed, quality of forage was not. It is possible that nutrient quality of forage items at created edges is less than that of natural edges. Gossman (2005) suggested that body condition of nekton was less at terraced edges than natural edges. Quality of forage in terraced ponds has not been evaluated for any other taxonomic groups.

Results for most species of conservation concern were inconclusive. This study was a community study and methods were not aimed at sampling specific species of concern. These species are rarer in nature, and were thus infrequently observed. Rather, analyses presented are meant to highlight trends and indicate areas were future research may be warranted. No species of concern had higher raw average density in terraced ponds. Four of the more common species of concern observed (Tricolored Heron, Gull-billed Tern, Common Moorhen, and Least Tern), like the other species in their foraging guilds, were significantly denser in terraced ponds. However, the statistical model used for two of these (Gull-billed Tern and Least Tern) violated the assumptions of ANOVA, and may not be valid. While inconclusive, these results suggest that most species of concern responded similarly to pond terracing as the rest of the foraging guild in which they were classified.

Bird density did not vary significantly with measured water quality variables. The lack of a water quality affect on density may result from the frequency at which water quality samples were taken. Waterbird densities commonly are shown to vary with water quality (Velasquez 1992, Halse et al. 1993, Nagarajan and Thiyagesan 1996). However, in my study water quality did not differ between pond types. Thus, it is reasonable that variation in bird density between restored and unrestored ponds is not explained by water quality. The only significant effect influencing bird density in ponds was the amount of available edge habitat. Fairbairn and Dinsmore (2001) preformed a similar analysis relating breeding waterbird densities to percent edge in prairie pothole wetlands in Iowa. Slopes relating percent edge to density for their analysis ranged from 0.95 to 310 depending on species (slope for my analysis was 1.06). R^2 for their analysis ranged from 0.2 to 0.74, but they preformed a multiple regression including many (possibly correlated) pond microhabitat cover variables. Birds in their study also were only surveyed once during the breeding season, so there was no variation due to migrational movements in their data. The R^2 for my regression was 0.153. I think this is a moderately high R^2 with biological significance, because main controls on bird density in Louisiana are likely to be factors outside the pond scale, such as location of breeding sites, continent-wide weather patterns, and particularly, migrational movements related to seasonal changes. Waterbirds in Louisiana are highly migratory (approximately 30% of observed species are usually classified as migrants). Variations in density with time have been seen in other studies of waterbirds in Louisiana (Spiller and Chabreck 1975). Thus, natural variation caused by these factors may mask variation in density resulting from pond scale variables. Further, although nekton density and SAV biomass did not influence bird density to a great enough degree to be detected by regression, birds preferentially used edge habitat, where foraging is more likely to be profitable. Maximum bird density and species richness in ponds where proportion of marsh edge is high and water:cover interspersion is maximized was also seen in studies by Weller and Spatcher (1965), Kaminski and Prince (1981), and Mack and

Flake (1980). My results in southern coastal marshes confirm the results of these northern freshwater marsh studies.

Other Options for Improving Waterbird Habitat in Marshes

Breeding success by waterbirds was not addressed in this study, but is an important determent in whether terraced marshes serves as population sources or sinks for resident birds (Erwin 2002). Creation of islands suitable for nesting rookeries would encourage breeding success of resident wading birds. Such islands should be isolated from mainland areas, small in size, and of an elevation to support adequate shrub cover (Greer et al. 1985, Bryan et al. 2003). Shorebirds, gulls, and terns prefer unvegetated areas for nesting, ideally high in shell substrate (Darnell and Smith 2004). Such areas also are important as loafing sites. Waterfowl, gulls, terns, and shorebirds at Vermilion and Rockefeller study sites frequently observed loafing on unvegetated terraces, which consisted of mixed mud and shell fragments. Maintenance of such unvegetated areas will promote waterbird abundance and diversity, but may increase pond turbidity.

Terraces differ from other restoration options because adding terraces manipulates the amount of edge habitat available. Most other marsh restoration options involve manipulating water depth and hydrologic inputs (Merino et al. 2005). The two restoration types can be combined to further improve waterbird habitat. Creating variation in water depth has been commonly seen to promote use of wetlands by multiple taxonomic groups of waterbirds (Parsons 2002, Taft et al. 2002, Bolduc and Afton 2004, Darnell and Smith 2004). Recommendations also are numerous on the management of water level to promote waterfowl, shorebirds, and desirable vegetation (Kadlec 1962, Harris and Marshall 1963, Vandervalk 1981, Twedt et al. 1998, Merino et al. 2005). Water depths less than 4 cm are ideal to promote shorebird use (Collazo et al. 2002). Shallow edges are most likely to become exposed and be used by shorebirds during late summer and early fall, a time when such habitat is critically needed and sparsely available for migrants (Twedt et al. 1998). Certainly, shorebirds in my study ponds only were seen on the rare occasions that exposed shallow margins were available, or during summer drought conditions following a semi-draw down at Rockefeller SWR. Shallow waters (10 to 19 cm) have been noted as ideal for wading birds because prey is both accessible and concentrated in shallow areas (Gawlik 2002). Taft et al. (2002) suggested that maximum waterbird density and diversity occurs on wetlands with average water depths of 10 to 20 cm and with topographic variability in water depths of 30 to 40 cm between deep and shallow zones. Water depths in my study ponds averaged much deeper than this (Table 5). Active water depth manipulation in most sites within the Chenier Plain is not possible, but can be mimicked by creating terraces with shallow slopes and broad shoulders.

CONCLUSION

Terraces increased the proportion of edge in ponds; the density of birds, nekton, and SAV were increased adjacent to marsh edges. Current construction designs may not increase the proportion of edge enough in all ponds to show whole pond effects for nekton and SAV. The amount of edge necessary to achieve pond level effects for SAV and nekton has not been evaluated Further increases in the proportion of edge can be accomplished by building terraces closer together, or by building them in curvilinear designs. Curvilinear designs may be more likely to slow water movements and encourage sediment settling than linear terrace designs. However, increases in spring and summer waterbird density were measurable at the whole-pond scale. Additionally, birds preferentially used marsh edge, where nekton and SAV were concentrated. Birds are highly mobile and visually oriented foragers, and can thus easily exploit areas were foraging is more likely to be profitable. Building terraces with broad shoulders and shallow slopes may further maximize waterbird density and species richness. This study only addressed spring and summer waterbird communities. Further study is warranted for wintering waterbird communities, many of whom are migrants that were not present during spring and summer.

The efficacy of terraces at slowing marsh erosion, preventing open water conversion, and encouraging emergent vegetation expansion has not been adequately evaluated. Causes of open water conversion in the Chenier Plain are not well understood. The majority of terraces are less than five years old, and determining whether terrace building is slowing or reversing marsh loss is difficult without monitoring over long-term scales. Some evidence suggests that terrace fields may be eroding with time under the effects of background wave action and hurricanes forces (personal observation following Hurricane Rita). Many wetland functions that depend on a well-developed soil organic matter layer take decades to return to undisturbed levels after a new disturbance. If constant repair of eroding terraces is necessary, such functions may never return to pre-disturbance conditions. Long term monitoring is necessary to determine whether terraces are sustainable or expanding.

CHAPTER 3: WATERBIRDS IN TERRACED VS UNTERRACED PONDS IN DURING WINTER

Pond terracing is a novel technique used ubiquitously in southwestern Louisiana's Chenier Plain to improve or repair certain functions in degrading coastal marshes. Louisiana contains the largest area of remaining coastal wetlands in continental United States (NOAA 1991), but wetlands in this area are also disappearing at a rapid rate (Barras et al. 2003). These marshes provide important habitat for large numbers of migratory and resident waterbirds (Keller et al. 1984, Greer et al. 1985, Myers et al. 1987, Martin and Lester 1990, Michot 1996), and their continued disappearance or degradation has the potential to greatly affect population declines locally and regionally. Pond terracing has become a key component of the restoration strategy for marshes in coastal southwestern Louisiana (Stead and Hill 2004). Thus, it is important to adequately evaluate terraced marshes as habitat for waterbirds.

The Chenier Plain is a unique environment in the southwestern portion of Louisiana. Marshes here are thought to be more stable than other marshes in the southeastern portion of the state (Barras et al. 2003), but marsh loss is still significant. The Chenier Plain lost an average of 16.3 km^2/year of wetlands from 1978 to 2000 (Barras et al. 2003). Marsh loss in the Chenier Plain results from two main causes: shoreline retreat resulting from Gulf Coast wave action (Byrnes et al. 1995) and interior marsh breakup. Pond terracing is intended to address this latter cause of marsh loss.

Interior marsh breakup results in the conversion of large areas of emergent marsh vegetation into shallow, open water ponds. This process is initiated by vegetation die-off in certain hot spots. Initial causes of this vegetation die-off in the Chenier Plain are not well understood. A variety of hypotheses have been set forth, including hydrologic alterations such as saltwater intrusion from canal dredging (Baumann and Turner 1990, Turner and Rao 1990, Gammill et al. 2001), geosyncline downwarping due to groundwater or oil and gas removal (Gosselink 1979), prolonged flooding and increased water depths due to various management projects (Gammill et al. 2001), toxic effects from industry runoff (Gosselink 1979), muskrat and nutria eat outs and/or ill-timed droughts (Bolduc and Afton 2003). Regardless of causes of initial die off, areas of open water often spread laterally via soil erosion a result of increased wave energy in larger open water areas. This phenomenon may be exacerbated by a variety of factors, including global sea level rise and sediment starvation caused by the channelization of the Mississippi and other rivers (Boesch et al. 1994, Turner 1997).

Terraces are intended to slow or reverse the process of open water conversion (Underwood et al. 1991, Steyer 1993, Rozas and Minello 2001). Terraces consist of long, discontinuous, narrow strips of created marsh. They are formed of dredge material stabilized by planting with emergent vegetation such as *Spartina alterniflora* (Underwood et al. 1991, Steyer 1993, Rozas and Minello 2001). Sediment for terrace building usually is taken from within pond bottoms, and is piled using backhoes, creating barrow pits within ponds.

Terraces are thought to function by providing a barrier to water movement. This is assumed to reduce wave energy and dampen the erosive force of water in large ponds (Underwood et al. 1991, Boesch et al. 1994). Additionally, slower moving water has decreased sediment carrying capacity, and may promote sediment deposition in terraced ponds. This could result in pond shallowing and increased soil fertility, creating a more hospitable environment for the establishment of emergent vegetation. This may also result in decreased water turbidity, possibly increasing submerged aquatic vegetation (SAV) production as well. Terracing also is thought to improve habitat by increasing the amount of edge (boundary between emergent vegetation and open water) within a pond (Rozas and Minello 2001). Shallow marsh edge is a highly productive zone for plants, nekton, and invertebrates (Gosselink 1979, Peterson and Turner 1994, Chesney et al. 2000) because it provides shallow low-energy area were detritus may accumulate, and also because vegetated edges may serve as a nekton nursery and refugia from large aquatic predators. Increasing the proportion of marsh edge has been noted to maximize waterbird density and diversity in northern freshwater marshes (Weller and Spatcher 1965, Mack and Flake 1980, Kaminski and Prince 1981, Murkin et al. 1982, Fairbairn and Dinsmore 2001) because it potentially increases production of forage items and maximizes habitat interspersion of cover and water. Thus, increasing the proportion of marsh edge should improve habitat quality for wildlife.

Seven previous studies have evaluated effects of terraces on coastal marsh functions. Steyer (1993) showed that terracing at Sabine National Wildlife Refuge increased primary productivity through the creation of emergent marsh (building of a terrace field) and subsequent expansion and colonization of emergents into adjacent open water areas. Cannaday (2006) and O'Connell (Chapter 2) examined terracing effects on SAV production at multiple sites, and concluded that terracing increased SAV directly adjacent to terraces edges. Cannaday (2006) also saw increased SAV biomass at whole-pond scales. Additionally, four studies examined terracing effects on nekton. These showed that nekton biomass was increased directly adjacent to terraced edges versus open water controls (Rozas and Minello 2001, Bush 2003, Gossman 2005, O'Connell Chapter 2), although there may be shifts in community composition (Rozas and Minello 2001, Bush 2003, Gossman 2005). Body condition of nekton also may be less at restored sites than unrestored sites, possibly because organic matter was less abundant at newly constructed terrace edges than at undisturbed edges (Gossman 2005). Two of these nekton studies (Rosas and Minello 2001, Bush 2003) were specific to only Sabine NWR, and thus extrapolating results to the entire Chenier Plain may be inappropriate.

Only one study has examined terrace effects on waterbirds (O'Connell Chapter 2), concluding that waterbird density and species richness was greater in terraced ponds than unterraced. This study examined spring and summer waterbird communities. However, many of Louisiana's waterbirds are migrants, and are only in coastal marshes during winter. This includes many species of migratory waterfowl (Bellrose 1980, Michot 1996, Esslinger and Wilson 2001). Further, two species classified as species of high concern by the Waterbird Conservation Council (Kushlan et al. 2002) are seen in Gulf Coast brackish marshes primarily in winter (American Bittern, and migratory King Rails). An additional four species of moderate concern occur primarily in winter (American White Pelican, Eared Grebe, Virginia Rail, and Common Loon). Effects of pond terracing on wintering waterbirds may differ from results seen in other seasons, and still needs to be evaluated.

I evaluated the quality of terraced ponds as wintering waterbird habitat by comparing waterbird density and species richness in restored and unrestored ponds. I also evaluated whether bird density in restored ponds varied by foraging guild. Additionally, to test assumptions in study design, and predictions of restoration managers, I compared water quality variables in restored and unrestored ponds. Finally, I evaluated which specific pond variables influenced waterbird density in terraced marshes.

METHODS

Study Area

My study sites were in coastal southwestern Louisiana within the Chenier Plain. This region of the Gulf Coast extends from Vermilion Bay, Louisiana, west to East Bay, Texas. It consists of shore-parallel, stranded inland beach ridges separated by broad areas of low elevation marsh. The Chenier Plain was formed by the long-shore westward transport of sediments from the Atchafalaya, Mississippi, and other rivers. These sediments were deposited in progradational mudflats along the shoreline. As the flats reached sufficient elevation, they were colonized by marsh vegetation. When the river delta shifted eastward, sediment was no longer deposited. Marine action gradually eroded the mudflats, and reworked the coarser grained sediments and mollusk shell, into higher elevation transgressive beach ridges, termed “cheniers.” This process was cyclical, as the rivers shifted from east to west many times, creating a series of linear chenier ridges separated by broad areas of marsh (Penland and Suter 1989).

Marsh in coastal Louisiana is classified into four types based on characteristic dominant plant communities (Penfound and Hathaway 1938, Chabreck 1970). These types are, in order of salinity, saline, brackish, intermediate, and fresh. These marsh types occur in bands parallel to the shoreline, with the saltiest zones closest to the gulf (Gosselink 1979). As of 1998, the Chenier Plain consisted of 135 km^2 saline marsh (4% of total marsh area), 803 km^2 brackish marsh (26%), 684 km^2 intermediate marsh (22%), and 1435 km^2 fresh marsh (47%) (Louisiana Coastal Wetlands Conservation and Restoration Task Force and Wetlands Conservation and Restoration Authority 1998).

I monitored ponds dominated by *Spartina patens* at study sites located throughout the Chenier plain. At each site, I monitored one terraced pond (treatment), and one nearby unterraced pond (control). Each pair is at a different site, and hydrologically distinct from the others (Figure 1).

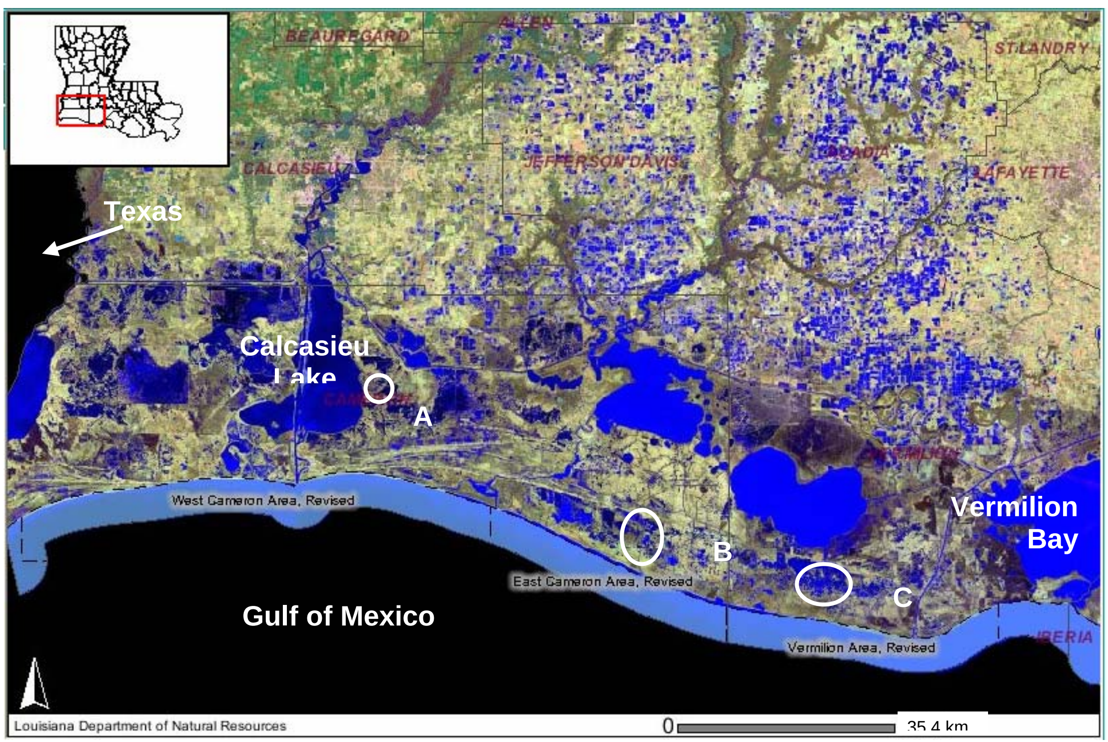


Figure 16. Louisiana’s Chenier Plain, showing locations of sites used in waterbird surveys during winter of 2006. A: Sweet Lake site; B: Rockefeller SWR site; C: Vermilion site

Site Selection

Study ponds were selected by first identifying marsh dominated by *Spartina patens*, which is an indicator of intermediate and brackish marsh types. From these, I excluded sites receiving major mineral sediment sources. Most wetlands in the Chenier plain do not receive large amounts of sediments from rivers, streams or bayous. Additionally, sites were picked only if they were known to have been emergent marsh prior to 1956. This last criteria was based on land change maps (1956-1990) created by Barras et al. (1994). Additionally, the terraces within sites also had to be mature enough to have established emergent vegetation. Finally, sites were included only if terraced and unterraced ponds were close to each other, of similar size, salinities, and under similar hydrologic regimes (i.e. undergoing the same water management scheme, or otherwise having similar water inputs). This left me with three sites, which were the only appropriate available pond pairs left in the Chenier Plain. All of these were included in the study.

Site Monitoring Effort Complications

Site monitoring throughout my study is not equal among all sites (Table 8). Sweet Lake was not surveyed 21 Jan. The landowner denied permission because waterfowl hunters were utilizing the area. Rockefeller SWR was not sampled on 28 Feb because of mechanical issues with our airboat. This site is impounded and requires an airboat to access it.

Table 8. Site monitoring effort for waterbird surveys conducted in winter of 2006, in the Chenier Plain, Louisiana.

Site	Date 1st sampled	Date last sampled	# Surveys Conducted
Rockefeller SWR	21 Jan2006	29 Mar 2006	5
Sweet Lake	28 Jan 2006	29 Mar 2006	5
Vermilion	21Jan 2006	29 March 2006	6
		Total # of Surveys:	6

Site Descriptions

<u>Sweet Lake-</u> These study ponds were located in Cameron Parish, LA, near Grosse Savanne hunting lodge (1730 Big Pasteur Rd., Lake Charles, LA 70607). The land was owned by Sweet Lake Gas and Oil Co., Miami Corporation, and Grosse Savanne Waterfowl & Wildlife Lodge. Calcasieu Lake borders the site to the west, and Sweet Lake borders it to the east. The terraced pond was directly north of the unterraced one. They were separated from each other by two spoil banks, which have a canal running between them (Figure 17). Both ponds were equidistant from Calcasieu Lake, the major source of saline water in the area. They were thus under similar hydrologic regimes. The terraces were constructed in 2001.

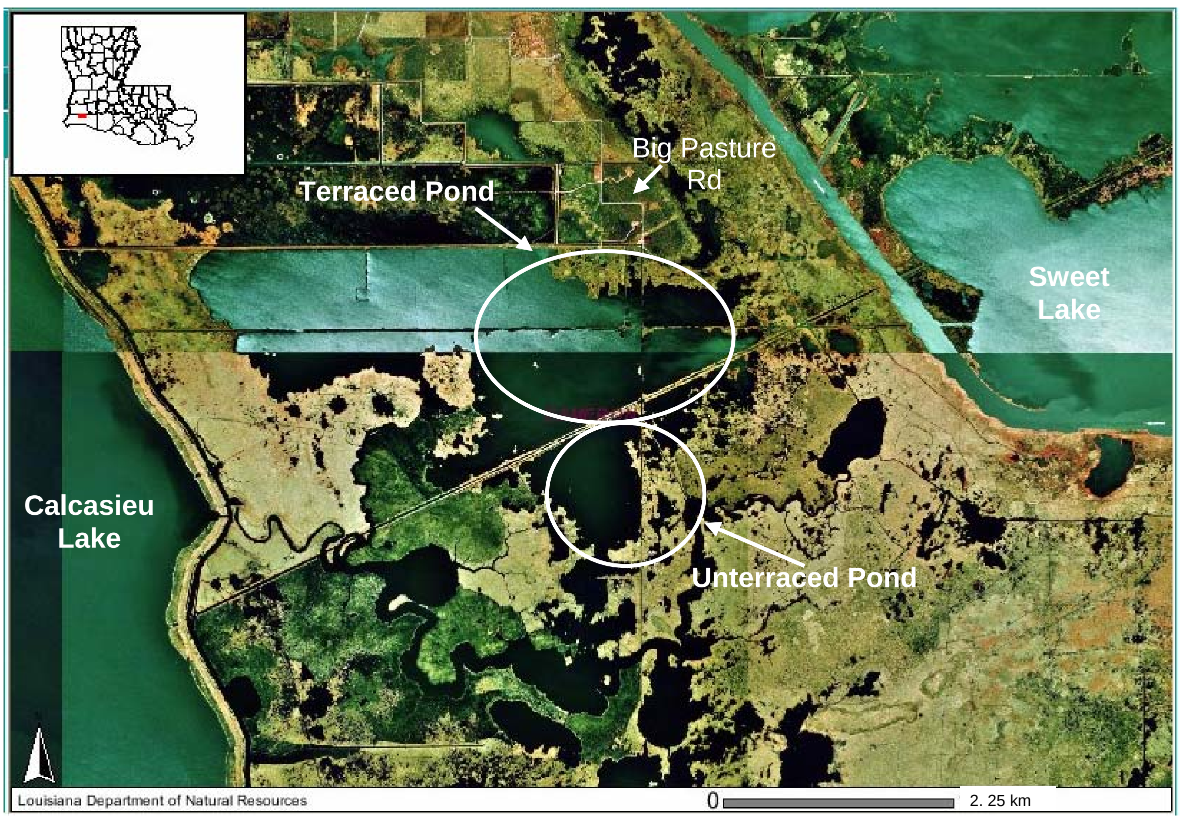


Figure 17. Sweet Lake survey ponds, Chenier Plain, Louisiana.

Rockefeller State Wildlife Refuge- The refuge is in southeastern Vermilion and southwestern Cameron Parishes (Hwy. 82, Grand Chenier, LA 70643). The land is owned by Louisiana Department of Wildlife and Fisheries. The refuge is situated between the Gulf of Mexico (to the south) and the Grand Chenier Ridge Complex, six miles inland (Melancon et al. 2000). The average elevation of the marsh in this area is 0.3 m above mean sea level (Chabreck 1960). The study ponds were located in Unit 4, an area of brackish impounded marsh actively managed for waterfowl. The terraced pond was directly north of the unterraced one (Figure 18). These terraces were constructed in 2002.

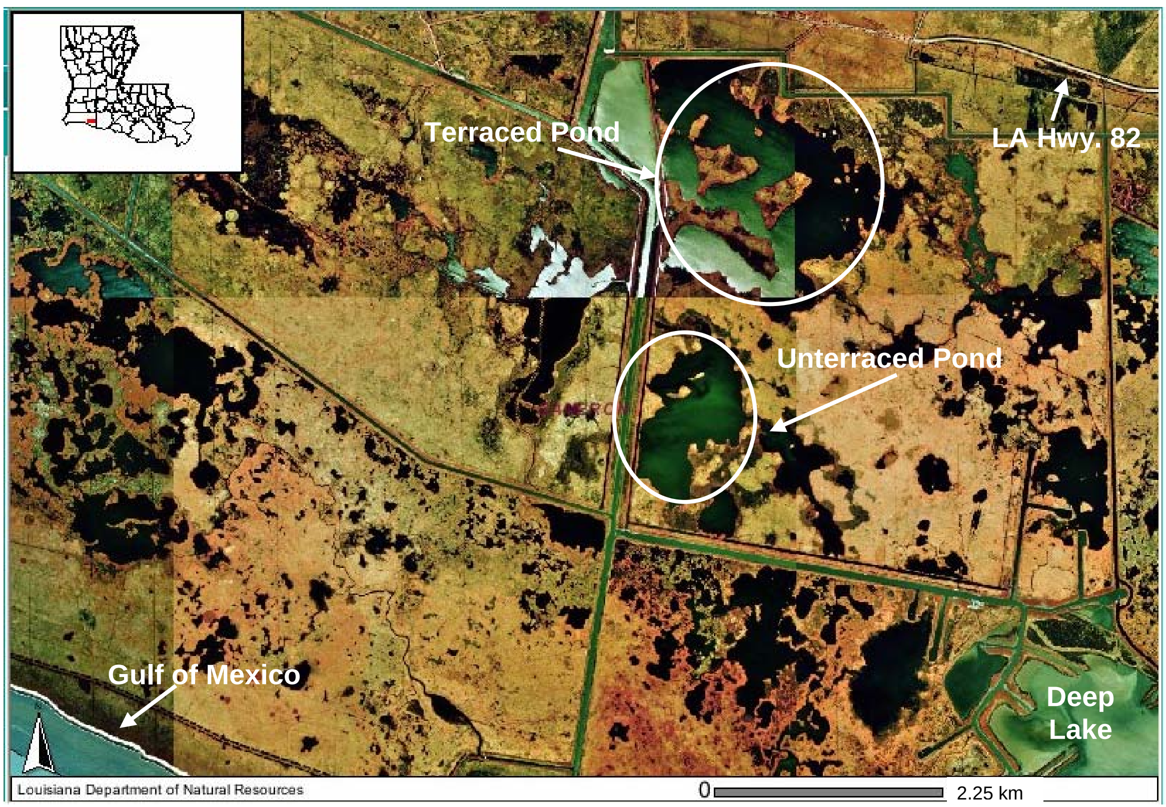


Figure 18. Rockefeller State Wildlife Refuge, Unit 4, Chenier Plain, Louisiana.

Vermilion- Two pond pairs were in an area of marsh south of Pecan Island, LA, owned by Vermilion Corporation (115 Tivoli St., Abbeville, LA 70510), in Vermilion Parish. The ponds are in an area of open, patchy marsh bordered by LA Hwy 82 to the north, Rockefeller SWR to the west, and the Gulf of Mexico to the south (Figure 19). These terraces were constructed in 2003.

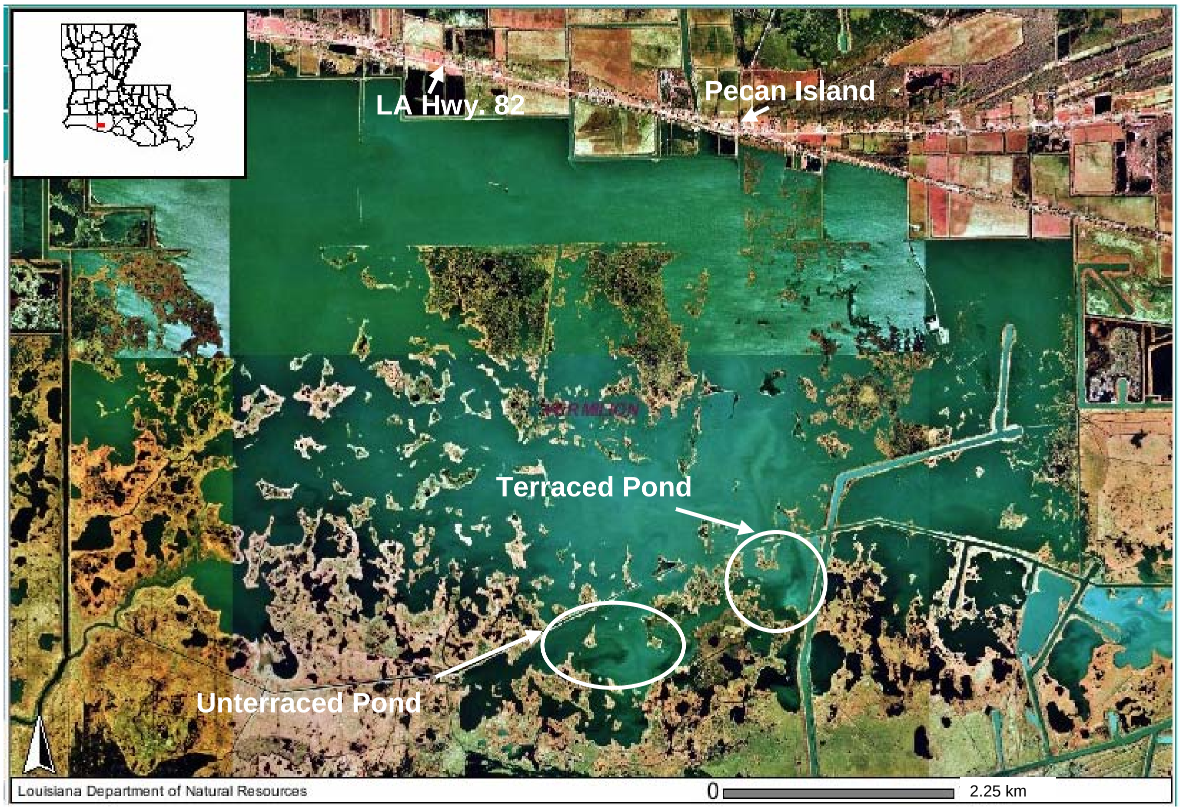


Figure 19.Vermilion Parish survey ponds, Chenier Plain, Louisiana.

Survey Methods

I designed my survey methods to maximize the number of plots, ponds, and sites that could be surveyed in a single day, due to lack of housing in the area following Hurricane Rita. Sites were prepped by marking the boundaries of two to three plots per pond using pvc pipe. Locations for plots were randomly chosen. Plot sizes ranged from 3 to 5 hectares. To conduct surveys, two observers and one boat driver were used, with sampling beginning during mid-day. During December 2005, I conducted test runs of survey methods and trained observers. During these test runs I generated appropriate techniques for reducing disturbance to birds from boat noise (approaching at low speeds and oblique angles, stopping at a distance, watching smaller discrete plots rather than running transects). Using these methods, flushing of birds was minimal. Some flushing did occur, but I believe that I successfully determined the location of birds prior to disturbance by the boat. In this manner, I was able to ensure that I counted only birds using my plots, rather than birds flushed up from adjacent marsh areas.

Two observers independently recorded observations, including species and number of birds present, distance of birds from observer, and when possible, behavior prior to disturbance (foraging, roosting or rafting) and microhabitat utilized (open water, mudflat, terraced edge, natural edge). After both observers had counted the plot, we then moved on immediately to count the next plot. Water quality measurements were collected (turbidity, salinity, conductivity, and water temperature) as well as wind speed and air temperature, once in the terraced pond, and once in the unterraced pond, greater than 25m from the edge. We used an EA-3010TWC anemometer (La Crosse Technology, 1116 South Oak Street, La Crescent, MN 55947 USA) to measure wind and air temperature. We used an YSI model 63 (Yellow Springs Instruments Inc., 1725 Brannum Lane, Yellow Springs, OH 45387 USA) to measure salinity, conductivity, and water temperature. Turbidity was measured using an Oakton Instruments T100 Turbidity Meter Kit, model WD-35635-00

(Oakton Instruments P.O. Box 5136, Vernon Hills, IL USA 60061), that was calibrated prior to each sampling effort. Turbidity samples were collected in undisturbed water. Fish and submerged aquatic vegetation data were not collected due to time constraints. All plots within a pond were surveyed during a single survey session. This method required only one or two days to survey all three sites. Sampling frequency was once every two weeks from 21 January 2006 to 29 March 2006.

Statistical Methods

Occasionally, surveys took multiple days to conduct. However, for the purpose of analysis, I assigned each survey a single date (the first day of surveying). I then converted the assigned date into the number of days since the beginning of the year.

Pond Characterization

I classified microhabitat types within ponds as described in Chapter 2. I used the nonparametric Wilcoxon Rank Sum test to compare proportion of available edge habitat between pond types. I used a logistic regression with pond type as the response variable and water quality variables as dependant factors to compare whether the water quality data differed between pond types.

Bird Analyses

I used an ANOVA with blocking on site and repeated measures over time to compare total bird density and species richness, between terraced (treatment) and unterraced (control) ponds (Table 9). Plots were considered replicates. I log transformed bird density to achieve normality.

Table 9. Experimental design of study to determine the effects of terraces on waterbird density and richness: an ANOVA with blocking on site and repeated measures. All site interactions were pooled into the error term *a priori*.

Factor	N	*df*	levels
Treatment (pond type)	2	1	Terraced, Unterraced
Date	6	5	Apr, May, Jun, Jul, Aug, Sept
Date*Treatment		5	
Site	3	2	Sweet Lake, Rockefeller, Vermilion
Total		35	

I grouped bird species into guilds by foraging method to evaluate if density of different guilds varied between pond types. To classify birds I generally used the foraging classifications of De Graaf et al (1985). My classification scheme differs somewhat from that of De Graaf et al's (1985). I categorized American White Pelican as divers, although they never dive. I categorized them as divers because the ponds are shallow and their long necks enable them to forage low in the water column. De Graaf et al. (1985) described Common Moorhens as both divers and dabblers, but I only observed them dabbling in our ponds, so I categorized them exclusively as dabblers. The resulting guilds are as follows:

1. Diving foragers: grebes, diving ducks, cormorants, American White Pelican
2. Wading foragers: herons, egrets, ibis, and Roseate Spoonbill
3. Shorebirds and other probers/surface arthropod gleaners: sandpipers, plovers, American Avocet, Black-Necked Stilt, and rails
4. Aerial foragers: terns, gulls, and Belted Kingfisher
5. Dabblers: dabbling ducks, Common Moorhen, American Coots, and Purple Gallinule

To tease apart differences in density in ponds by birds with different foraging methods, I compared guild density between pond types. In all cases, log transformations were necessary to obtain normal response variables.

A number of species of conservation concern occur in Louisiana's coastal brackish marshes. It is possible that pond terracing effects habitat for species of concern differently than it effects habitat for other species. For this analysis, I used the conservation classifications proposed by the Waterbird Conservation Council (Kushlan et al. 2002). These include 13 Gulf Coast species that are classified as species of high concern (Black

Skimmer, Least Tern, Little Blue Heron, Snowy Egret, Tricolored Heron, Gull-billed Tern, Roseate Tern, Pied-billed Grebe, Purple Gallinule, American Bittern, King Rail). It additionally includes 14 Gulf Coast species classified as species of moderate concern (American White Pelican, Forster's Tern, Anhinga, Neotropic Cormorant, Reddish Egret, Roseate Spoonbill, White Ibis, Black-crowned Night-Heron, Eared Grebe, Royal Tern, Clapper Rail, Virginia Rail, Common Moorhen, Common Loon).

To be included in analysis, a species had to have been observed on at least three separate occasions. I compared density of each these species of concern between pond types, using a repeated measures ANOVA with blocking on site (Table 9). Residuals of results were examined, and log transformations were necessary to improve normality of response variables and homogeneity of variances. When significant pond type by time interactions were seen, six post hoc tests with tukey adjustments were used to compare responses between pond types on each survey date.

Additionally, I used a backwards stepwise regression to determine whether any measured variables could explain variation in bird density without including pond type in the model. Potential explanatory variables water temperature, turbidity, salinity, conductivity, air temperature, wind speed, and proportion of microhabitat types (edge or open) in ponds.

RESULTS

Pond Characterization

Water quality (turbidity, salinity, conductivity, water temperature) did not differ between terraced and unterraced ponds for any variable measured (Table 10).

Table 10. Mean water quality (+/- one se) for terraced and unterraced ponds, winter of 2006, Chenier Plain, Louisiana. Water quality was not significantly different between pond types.

Variable	Terraced Pond	Unterraced Pond
Turbidity	87.1 +/- 34.4	107.1 +/- 31.4
Salinity	7.8 +/- 0.8	8.1 +/- 0.8
Conductivity	315.3 +/- 302.6	13.3 +/- 1.2
Water temperature	18.0 +/- 0.9	17.2 +/- 0.7

Ponds averaged 80 hectares (range 9 to 470 hectares). Adding terraces to ponds increased the proportion of pond edge habitat (Chapter 2, Table 6).Terraced ponds had more edge habitat (40.0%, se = 3.2%) than did unterraced ponds (8.2%, se = 0.8%), when terraced edge and natural edge were combined into one marsh edge category ($p < 0.0001$).

Bird Results

Figure 20 presents the raw data for waterbirds for all surveys. To give a visual representation of species richness, each wedge represents relative frequency for species that were seen more than twice. Birds seen less often are lumped together into "other." The area of the pies is proportional to total bird density from all surveys.

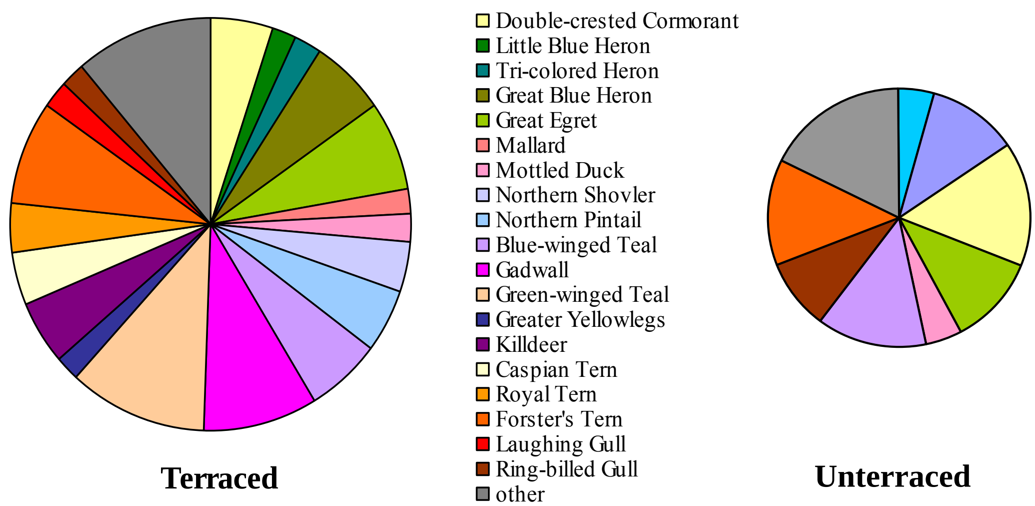


Figure 20. Raw bird density and species richness in terraced vs. unterraced ponds, summed over all surveys, winter of 2006, Chenier Plain, Louisiana. Size of charts is proportionate to density.

Raw mean bird density (birds/hectare) was 5.0 (se = 1.3) in terraced ponds and 1.5 (se = 0.2) in unterraced ponds, a 70 % increase. Raw mean bird species richness was 2.7 (se = 0.35) in terraced ponds and 1.3 (se = 0.02) in unterraced ponds, a 50% increase. After log transformations, bird density and species richness were significantly greater in terraced ponds (Figure 22, $F_{1, 57} = 14.38$, $p = 0.0004$ and $F_{1, 57} = 15.05$, $p = 0.0003$ respectively).

Guild Response

Figure 21 shows bird density and relative frequency by guild. Size of pies is proportional to total bird density. When pond types are combined, guilds were, in order of lowest average density (birds/hectare) to highest: shorebird (0.10, se = 0.05), waders (0.11, se = 0.03), divers (0.16, se = 0.07), aerialists (0.22, se = 0.05), and dabblers (2.64, se = 0.73).

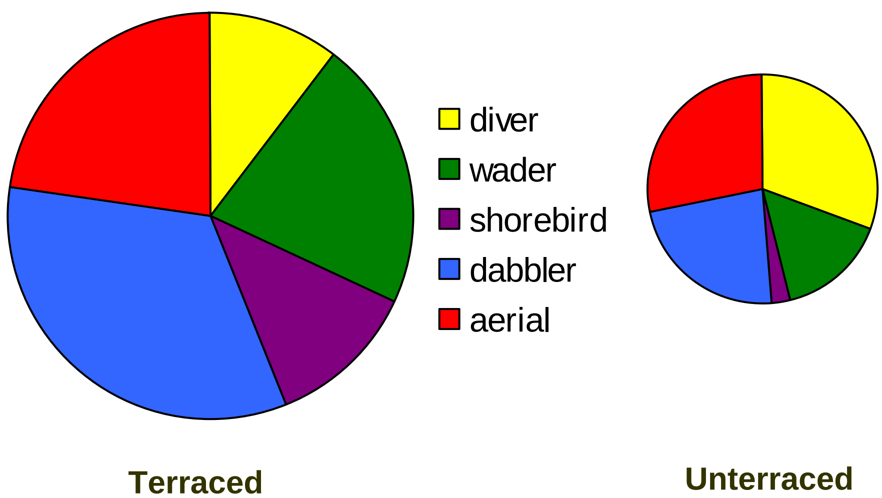


Figure 21. Raw bird density and relative frequency of foraging guilds in terraced and unterraced ponds, winter of 2006, Chenier Plain, Louisiana. The area of the pies is proportionate to density.

Dabbling and wading birds were more dense in terraced ponds than in unterraced ponds (Figure 22, $F_{1, 57} = 12.3$, $p = 0.0009$ and $F_{1, 57} = 7.83$, $p = 0.007$ respectively), but residuals were not normal despite attempts at transformations. Mean log of dabbler density (dabblers/hectare) was 0.94, se = 0.19 (raw average = 4.19, se = 1.26) in terraced ponds, and 0.24, se = 0.19, (raw average = 0.95, se = 0.61) in unterraced ponds. Mean log of wader density was 0.13, se = 0.031, (raw average = 0.17, se = 0.047) in terraced ponds, and 0.037, se = 0.033, (raw average = 0.044, se = 0.13) in unterraced ponds. Aerialist were relatively abundant, but densities did not differ between pond types (terraced raw average = 0.25, se = 0.075; unterraced raw average = 0.19, se = 0.58). Shorebird (plus rails) and diving birds were least abundant, and densities did not differ between pond types (shorebird raw average: terraced = 0.14, se = 0.08, unterraced = 0.045, se = 0.045; diver raw average: terraced = 0.08, se = 0.04, unterraced = 0.25 se = 0.08).

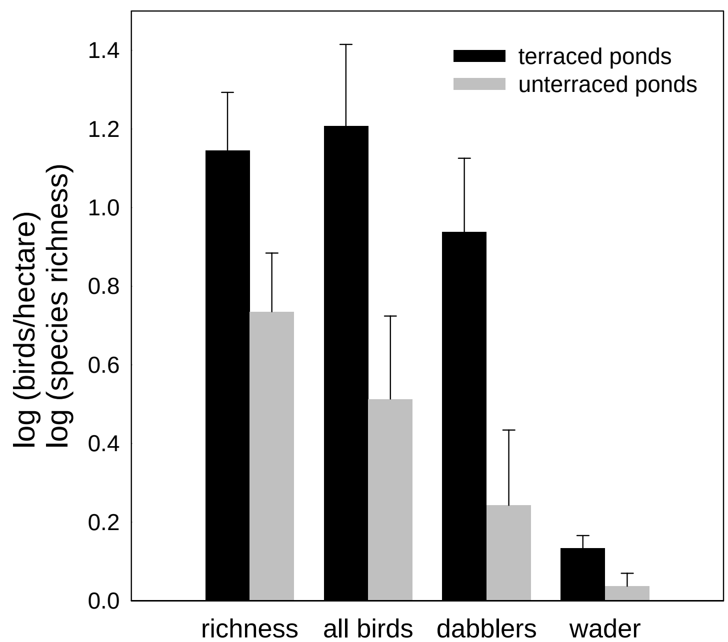


Figure 22. Log of bird species richness and log of density for all birds, dabblers, and waders in terraced and unterraced ponds (mean +/- se), in winter of 2006, Chenier Plain, Louisiana.

Species of Concern Response

Eleven species of conservation concern were observed in my study ponds during winter (Table 11). Few species were observed commonly, and most species did not differ significantly between pond types. Log transformations improved normality of model residuals, however complete normality of residuals was never achieved. Raw means for most species were greater in terraced ponds. Two species, Snowy Egret and Pied-Billed Grebe, had higher raw means in unterraced ponds. However, only one Snowy Egret was ever observed. Two species that were observed frequently did not differ significantly in density between pond types (Pied-billed Grebe and Forster's Tern). Three other species were frequently observed (Royal Tern, Little Blue Heron, and Tricolored Heron). Royal Tern's were significantly denser in terraced ponds. Little Blue Heron's and Tricolored Herons did not differ significantly between pond types, but there was a nonsiginificant trend towards higher density in terraced ponds.

Finally, bird densities did not vary with any measured water quality variables, air temperature, or wind, but did vary with proportion of edge habitat in ponds. Ponds with more edge had higher bird densities (Figure 23, $F_{1, 69} = 17.45$, $p < .0001$).

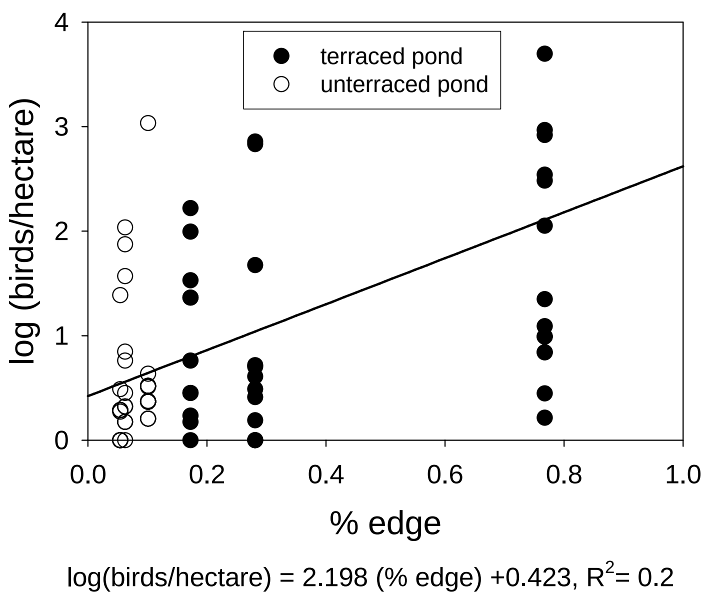


Figure 23. Relationship of log mean waterbird density and percent of vegetated marsh edge available in ponds, winter of 2006, Chenier Plain, Louisiana.

Table 11. Results for species of conservation concern, with raw means of density (birds/hectare) for pond type and standard errors. No. is the total number of separate occasions this species was seen. Sig. Effect is whether significant results were for pond type, pond by time interactions, or no significant effects were seen. F-statistic and p-value are for the significant effect listed. Normal is whether normality was achieved for model residuals after the response variable was log transformed.

Bird	Terraced	SE	Unterraced	SE	No.	Sig. Effect	F- statistic	p-value	Normal
Little Blue Heron	0.01	0.008	0	0	2	Pond type trend	$F_{1, 57} = 2.96$	0.09	No
Snowy Egret	0	0	0.007	0.007	1	None			No
Tricolored Heron	0.01	0.008	0	0	2	Pond type trend	$F_{1, 57} = 2.82$	0.099	No
Gull-billed Tern	0.005	0.005	0	0	1	None			No
Royal Tern	0.02	0.01	0	0	4	Pond type	$F_{1, 57} = 7.07$	0.01	No
Pied-Billed Grebe	0.01	0.012	0.04	0.02	6	None			No
Forster’s Tern	0.14	0.07	0.14	0.08	14	None			No
Neotropic Cormorant	0.02	0.02	0	0	1	None			No
Roseate Spoonbill	0.02	0.02	0	0	1	None			No
White Ibis	0.02	0.02	0	0	1	None			No
Eared Grebe	0.007	0.007	0	0	1	None			No

DISCUSSION

Water quality (water temperature, salinity, conductivity, and turbidity) did not differ significantly between pond types. These results confirm assumptions that control ponds chosen were similar to terraced ponds in hydrology. Further, contrary to predictions about terrace effects, turbidity was not altered from unrestored conditions. Reduction of water turbidity often is cited as the mechanism through which submerged aquatics will be promoted by pond terracing. This is similar to results seen by O'Connell (Chapter 2). A more frequent sampling regime might have revealed a difference in water quality. However, no previous study has ever conclusively documented turbidity reductions in terraced ponds and the extent to which sediment settling occurs is largely unevaluated. Still, all terraces studied were less than five years old. Terracing is a novel restoration tool, and nearly all terracing projects are less than five years old (Stead and Hill 2004). Terraces probably function as patches of created marsh within ponds. It has been observed in other marsh creation projects that many system functions take decades to return to pre-disturbance conditions at newly created sites (Craft et al. 1999, Zheng et al. 2004). It is possible that terraces in these ponds were not mature enough to have developed a turbidity reduction function.

Additionally, O'Connell (Chapter 2) argued that lateral expansion by emergents adjacent to terraced and natural edges in terraced marsh is an important indicator of whether terraces are reversing open water conversion. While reversal of open water conversion as a result of pond terracing was hypothesized by early terrace proponents (Underwood et al. 1991, Steyer 1993), it has never been conclusively proved. Although not measured directly, expansion of emergents into open water adjacent to terrace edges at any site during summer was not visually obvious (Chapter 2), and this additionally was not observed during the winter. Without the encouragement of emergent production laterally and vertically, over the long-term terrace effects are likely to be transitory.

Hurricane Rita hit the Chenier Plain in September 2005 (just prior to this study), causing near total destruction of coastal development and substantial upheaval in adjacent marshes. For example, in my study ponds numerous “marsh balls” of uprooted *Spartina patens* were seen deposited in many previously open water areas. Salinities also were elevated from those seen during summer sampling in the same region (see Chapter 2), probably due to unflushed storm over-wash. Additionally, much of the emergent vegetation on terraces and in natural marsh appeared brown, though this may have been a result of winter dormancy rather than salt-burn from storm surge. Storm related erosion was not obvious in my study ponds. Few measurements were made, but there was no visually apparent alteration of terrace widths or of surrounding natural marsh edge morphology directly following the hurricane. Additionally the sides of terrace edges did not appear undercut. However, I revisited Rockefeller SWR in late spring of 2006, after all surveys for this study had been completed. At this time, the terraces at this impounded site were vastly degraded, perhaps as a result of sustained elevations in salinity causing vegetation kills. Long-term effects of hurricanes on terraced marsh has yet to be determined.

I have no mechanism for investigating to what extent the hurricane influenced wintering waterbird populations in the Chenier Plain during this study. I have no pre-hurricane data for the wintering period, and direct mortality was not investigated. However, species observed did not seem atypical for coastal Louisiana. Many of the winter migrants were not present when the hurricane struck, and may have been hardly affected by it.

Terracing increased the proportion of marsh edge habitat in ponds, with resulting increases in the abundance of wildlife. Birds responded to pond terracing at both the microhabitat and whole-pond scales. Bird density and species richness were greater in terraced ponds than in unterraced ponds. There was no significant interaction with time. This is likely because individual surveys were close together in time and over only one season (winter), such that species composition was similar throughout the entire study period.

Foraging guilds varied in their response to pond terracing. Waders and dabblers were both consistently denser in terraced ponds. Shorebirds, aerialists, and divers showed no significant treatment effects. Dabbling foragers likely benefited from pond terracing because of increases in SAV directly adjacent to terrace edge (O'Connell Chapter 2). Dabblers forage for SAV, SAV seeds, and aquatic invertebrates among SAV (Ehrlich et al. 1988). SAV also provides structure, refuge, and forage for nekton (Rozas and Odum 1988, Castellanos and Rozas 2001), and thus may provide a highly profitable foraging area omnivorous and piscivorous birds. Aerialist foragers were more abundant than were waders, but did not differ in their density between pond types during winter. Interestingly, the spring/summer study (Chapter 2) detected differences in density between pond types for aerial foragers. This may be because aerialists behave differently during different seasons, or it may be an artifact of either hurricane effects or sampling methods. Conversely, diving foragers results were similar to those seen during the spring/summer study. Diving foragers were the only guild with higher raw densities in unterraced ponds, but density did not differ significantly between pond types All of these ponds are shallow (usually less than 1m), and although diving foragers do use them, they probably don't constitute important habitat for diving foragers. Shorebirds were least abundant, and showed no significant treatment effects over the winter. This may be because water depths in winter are too deep in these ponds to support high numbers of shorebirds. Water depths may also have been deeper than usual as a result of undrained storm surge. However, although some species of shorebirds winter in Louisiana, they are generally more abundant during migration in spring and fall.

Results for most species of conservation concern were inconclusive. This study was a community study and methods were not aimed at sampling specific species of concern. These species are rarer in nature, and were thus infrequently observed. Rather, analyses presented are meant to highlight trends and indicate areas were future research may be warranted. Most species of concern had higher raw average density in terraced ponds. The most frequently observed exception to this was Pied-Billed Grebes, who were observed more often in unterraced ponds. All diving foragers, such as grebes, did not differ significantly in density between ponds, but had higher raw densities in unterraced ponds. Additionally, Forster's Terns, like other aerial foragers, were frequently observed, had higher raw means in terraced ponds, and yet did not differ in density between pond types. Two wading bird species of concern also had similar responses to the rest of their foraging guild, in that they tended to have higher density in terraced ponds, although this trend was not significant ($p = 0.09$ for both species). While inconclusive, these results suggest that most species of concern responded similarly to pond terracing as the rest of the foraging guild in which they were classified. The exception to this was Royal Terns, whom, unlike other aerialist foragers, were significantly denser in terraced ponds.

Bird density did not vary significantly with measured water quality variables. The lack of a water quality affect on density may result from the frequency at which water quality samples were taken (once every two weeks). This same result was seen during spring and summer at these same sites (O'Connell Chapter 2). Sampling frequency for that study was once a month. Waterbird densities commonly are shown to vary with water quality (Velasquez 1992, Halse et al. 1993, Nagarajan and Thiyagesan 1996). However, in my study, water quality did not differ between pond types. Thus, it is reasonable that variation in bird density between restored and unrestored ponds is not explained by water quality. The only significant effect influencing bird density in ponds was the amount of available edge habitat. The R^2 for this regression was 0.2. This is similar to results seen for the same analysis ($R^2 = 0.15$) during spring and summer in these sites (O'Connell Chapter 2). I feel this is moderately high R^2 with biological significance because main controls on bird density in the region are likely to be factors outside the pond scale, such as location of breeding sites, continent-wide weather patterns, and migrational movements related to seasonal changes. Maximum bird density and species richness in ponds where proportion of marsh edge is high and water:cover interspersion is maximized also was seen in studies by Weller and Spatcher (1965), Kaminski and Prince (1981), Mack and Flake (1980) and Fairbairn and Dinsmore (2001). My results regarding wintering waterbirds confirm the results of these studies in northern freshwater marshes, and supports results seen for southern coastal marshes during spring and summer.

Recommendations for Future Wetland Restorations Aimed At Improving Waterbird Habitat Quality

Shorebirds, gulls, and terns prefer unvegetated areas for loafing sites (Darnell and Smith 2004). Waterfowl, gulls, terns, and shorebirds at Vermilion and Rockefeller study sites were frequently observed loafing on unvegetated terraces, which consisted of mixed mud and shell fragments. Maintenance of such areas will promote waterbird abundance and diversity, but may increase pond turbidity.

Pond terracing differs from other management techniques because adding terraces manipulates the amount of edge habitat available. Most other marsh management options involve manipulating water depths and hydrologic inputs (Merino et al. 2005). The two management types can be combined to further improve waterbird habitat. Variation in water depth has been commonly seen to promote use of wetlands by multiple taxonomic groups of waterbirds (Parsons 2002, Taft et al. 2002, Bolduc and Afton 2004, Darnell and Smith 2004). Literature is voluminous on the management of water level to promote waterfowl, shorebirds, and desirable vegetation (Kadlec 1962, Harris and Marshall 1963, Vandervalk 1981, Twedt et al. 1998). Water depths less than 4 cm are ideal to promote shorebird use (Collazo et al. 2002). Shallow edges are most likely to become exposed and be used by shorebirds during late summer and early fall, a time when such habitat is critically needed and sparsely available for migrants (Twedt et al. 1998). Certainly, shorebirds in my study ponds only were seen on the rare occasions that exposed shallow margins were available, and were less dense during winter than summer, when water was deeper. Shallow waters (10 to 19 cm) have been noted as ideal for wading birds because prey is both accessible and concentrated in shallow areas (Gawlik 2002). Taft et al. (2002) suggested that maximum waterbird density and diversity occurs on wetlands with average water depths of 10–20 cm and with topographic variability in water depths of 30–40 cm between deep and shallow zones. Water depths in my study ponds averaged much deeper than this (Table 5). Variation in water depth can be created by constructing abundant habitat with wide, shallow, sloping edges. Such edges should dry and reflood with seasonal changes in hydrologic inputs.

Water depth also can be actively manipulated in marshes by impounding sites and actively pumping water, or installing weirs. Active water depth manipulation in most sites within the Chenier Plain is not possible, but fixed crest weirs have been used in coastal Louisiana to stabilize water levels and salinities in marshes. Weirs hold water in ponds when water levels would otherwise be very low, i.e. when strong northerly winds are pushing water out of marshes during the winter months. The effects of weirs on marsh functions have been reviewed by Chabreck and Hoffpauer (1962) and Burleigh (1966). Spiller and Chabreck (1975) found that weired ponds contained deeper water, and four times more ducks and coots during December and February than ponds not influenced by weirs. Other waterbird populations were 22% denser in weired ponds than unweired during December. During months when water levels in nonweired ponds were similar to weired ponds, waterbird densities were similar in the two marsh types. However, terraced marshes in winter had 77% more waterfowl and 70% more waterbirds than did unterraced marshes.

Terraced ponds also had 70% greater bird densities during spring and summer (O'Connell Chapter 2), and may thus be a more effective than weirs at improving waterbird habitat, depending on local conditions and management objectives.

CONCLUSION

Terraces increased the proportion of edge in ponds and the density of birds by 70% in winter. Further improvements in waterbird habitat quality may be achieved by building terraces with shallower slopes, and by including nesting habitat in construction designs. The efficacy of terraces at slowing marsh erosion, preventing open water conversion, and encouraging emergent vegetation expansion has not been adequately evaluated. Causes of open water conversion in the Chenier Plain are not well understood. The majority of terraces are less than five years old, and determining if terrace building is revering marsh loss is difficult without monitoring over long-term scales. Some evidence suggests that terrace fields may be eroding with time under the effects of background wave action and hurricanes forces (personal observation following Hurricane Rita). Many wetland functions that depend on the adequate soil organic matter development take decades to return to undisturbed levels after a new disturbance. If constant repair of eroding terraces is necessary, such functions may never return to pre-disturbance conditions. Long term monitoring is necessary to determine if terraces are sustainable and/or expanding.

CHAPTER 4: USE OF PHOTOSENSITIVE CAMERAS TO CONDUCT WATERBIRD SURVEYS

Waterbird populations can be censused in many ways. Some methods may be more efficient or accurate than others. The original project proposal for my thesis research suggested using camera systems to survey waterbirds. As preliminary field trials were conducted, if became obvious that the efficiency of this method needed to be evaluated. Traditional waterbird survey methods involve observers equipped with high-powered optics (spotting scopes and binoculars) using some standardized method to survey plots or transects within habitats of interest (Emlen 1971, Ralph and Scott 1981, Frederick et al. 1996, Nichols et al. 2000, Bart and Earnst 2002, Rosenstock et al. 2002, Watson 2003). The alternative method proposed for my thesis research involved using automatic camera systems to record birds on plots without observers present. In theory, this alternative method has several advantages. It may help to eliminate variability resulting from differences in skill among observers and to eliminate disturbance caused by human observers. If manpower is limited and multiple camera systems are available, then simultaneous observations of all habitats of interest is an additional benefit when using camera systems. Simultaneous observations should be superior to consecutive observations because of variability in wildlife numbers caused by animal movement associated with weather, circadian rhythms, and consecutive observer disturbance. Further, if manpower is limited and sites are remote, then cameras programmed to record using automatic timers could potentially generate more survey hours per plot than a live observer. Also, video recordings of plots can be played back in the lab thereby permitting exact counts that would potentially provide increased accuracy.

There are also several potential disadvantages to using automatic camera systems. If live observers aren't present to operate them, they can not be aimed, focused, or zoomed onto any particular animal, and poor image quality may result. Also, camera field of view can not be moved, so plots observed by cameras are smaller than those surveyed by human observers. Reliability may also be a problem. Of necessity, automatic camera systems are composed of many complicated electrical parts, each of which may fail, be lost by technicians, or be improperly deployed. In addition, if camera surveys are to be more efficient than live observer surveys, then it must take less time to set up and arm all the parts of the system than it would to conduct a bird survey using live observers.

To evaluate the efficiency and accuracy of this survey method, I used solar-powered camera systems with automatic, photosensitive timers to survey waterbirds and I then compared the results of these surveys to those conducted by observers from the same general time period.

METHODS

Study Area

All surveys were of Rockefeller State Wildlife Refuge, Louisiana, in Unit 4 (lat 29° 41' 2.1"N, long 92° 45' 28.4"W). This unit consists of impounded brackish marsh dominated by *Spartina patens* and two large, shallow (less than 1m in depth) open water brackish ponds, which are commonly used by waterbirds. All surveys were all conducted within the northern-most open water pond.

Camera Surveys

The components of the camera systems were built by Sandpiper Technologies (535 W. Yosemite Avenue, Manteca, CA 95337, USA). Each system consisted of the following items (Figure 24):

A) Sentinel^{TM} All-weather Video Surveillance system (VCR protected in a pelican case)

B) Sunrise/Sunset Time-lapse Video Recording Controller with on board Vertical Intefreated Time Code (VITC) Generator (photosensitive timer capable of stamping time and date onto VHS video)

C) STI Solar Charger 100 (100-watt solar panel)

D) Deep-cycle marine battery (115 amp hours)

E) PicoCam TeleZoom HWB-2 5A6 (Color surveillance Camera)

The photosensitive timers allowed power to the systems for two hours after sunrise, and two before sunset. The camera systems were programmed to record during these hours for six consecutive days. The systems were deployed on ten-foot tall aluminum ladders, placed in the center of open water ponds. These ladders were modified to hold the equipment by the addition of heavy wooden shelves and stabilizing supports. Cameras were deployed in randomly chosen locations within ponds. They were aimed towards the nearest emergent vegetation, on either a terrace or natural marsh edge, which ever was closer. To protect the equipment, lightning rods, constructed from copper pipe, were placed nearby (Figure 24).

Figure 24. Camera system (camera, solar panel, VCR, and timer) deployed in an unterraced pond in Rockefeller's Unit 4, winter of 2005, Chenier Plain, Louisiana. Tripod made of ten-foot ladder with attached 6 ft. shelf. Lightning rod made of copper and aluminum pipe is in foreground.

Preliminary trials were conducted to determine maximum camera plot size (35 to $50m^2$). I conducted trial waterbird surveys using these systems on two occasions over the winter of 2004 to 2005. Four cameras were deployed on each occasion, two in the terraced pond, and two in the unterraced pond. Systems were left deployed for one week. Video were then collected and digitized in the lab using a Canopus MVR 1000 Real time MPEG encoder (Canopus Corporation, 711 Charcot Avenue, San Jose, CA 95131, U.S.A.) with Canopus Mediacruise Digital AV Control System software (version 2.23.000).

Observer Surveys

Observers arrived at plots at dawn equipped with spotting scopes (Eagle Optics Raven 78mm straight scope, 20-60x power, Eagle Optics, 2120 W. Greenview Dr., Middleton, WI 53562, USA), and binoculars (Nikon Monarch ATB 10 x 42, Nikon Vision CO., LTD. 3-25, Futaba 1-chrome, Shinagawa-ku, Tokoyo 142-0043, Japan).

Observers sat in the nearest emergent vegetation, using camouflage netting for additional cover. Observers allowed a fifteen-minute settling period following their arrival on plots and then surveyed birds for a ninety-minute period. Observer plots were 120,000 m^2, which preliminary trials suggested was the maximum possible plot size observer could monitor.

Data from real-time observations and that extrapolated from video was recorded in the same fashion. At 15 min. intervals, bird abundance and diversity for all birds using the pond were recorded. Fly-bys were not recorded. To obtain a conservative estimate for bird abundance, only the maximum number for a species seen during any one 15-minute interval was kept for statistical analyses.

Statistical Methods

Four cameras were deployed on two occasions, yielding eight possible camera surveys. On the first deployment, only one camera successfully recorded. On the second deployment, two cameras successfully recorded, yielding three camera surveys total. To compare camera surveys with those made by live observers, three live observer surveys from around the same time period, and where possible, from the same location, were chosen (Table 12). An ANOVA comparing total birds observed between survey types (camera or live), with blocking on plot was preformed. To obtain normality of residuals, a log transformation was preformed on total birds.

RESULTS AND DISCUSSION

Several attempts at using the cameras to efficiently conduct bird surveys failed.

Table 12. Date, survey type and plot used for camera vs. live observer waterbird survey method comparison, winter of 2004 to 2005, Chenier Plain, Louisiana. Plots with the same label are in the same location, but plots for live observers are larger than those used for camera surveys.

Date	Type	Plot
12/11/2004	Camera	RT1
1/15/2005	Camera	RT1
1/15/2005	Camera	RT2
1/15/2005	Live	RT1
1/15/2005	Live	RT2
2/11/2005	Live	RT3

Equipment was bulky and difficult to handle, time consuming to deploy (at least one full day required per site), and successfully recorded only sporadically. Field of view was small (35 to 50m^2) compared with that available to a live observer (120,000m^2), and birds rarely swam in front of the camera, even when hundreds of waterfowl were present and available to do so. When birds did swim in front of the camera, image quality was such that birds could often not be identified to species (Figure 25).

Figure 25. Still image from camera. Camera was deployed in a pond during winter of 2005, at Rockefeller SWR, Chenier Plain, Louisiana. Camera was aimed to face the nearest emergent vegetation. The dark blotches in the open water close to the emergent vegetation probably are ducks.

Bird abundance counted by observers and by photosensitive cameras differed significantly ($F_{1, 2}$ = 21.66, p = 0 .0432). Mean log of bird abundance was 0.46 (standard error = 0.45) for camera surveys, and 3.43 (standard error = 0.45) for surveys using live observers. Moreover, the mean number of birds observed by cameras did not differ significantly from zero. Yet during this period, hundreds of birds were present in the pond (mostly waterfowl).

Observers were better than cameras at detecting birds because they used high quality, high-powered optics (binoculars and spotting scopes). Such optics are more capable of observing large areas, focusing as necessary, and delivering clear, bright images, especially under low light conditions. I abandoned the camera method of surveying as inefficient and developed survey methods using only observers.

CHAPTER 4: PROJECT CONCLUSIONS

My results indicate that marsh terracing has important influences on wildlife densities in coastal marshes. This project contained two studies, one that examined terraced marsh as habitat for spring and summer waterbird communities, and one that examined terraced marsh as habitat for winter waterbird communities. Methods for these two studies differed, but results were similar, as were the overall conclusions.

Terracing did not change water depth or turbidity in ponds at coarse scales. Further, no study has adequately evaluated whether terraces slow or reverse marsh loss in the Chenier Plain, though they continue to be built for this purpose. However, pond terracing significantly increased the proportion of vegetated marsh edge in ponds. It is through this mechanism that pond terracing improved habitat quality for waterbirds. Nekton and submerged aquatic vegetation were denser at marsh edge (both natural and terraced edges) during spring and summer. However terraced ponds did not contain significantly more nekton or SAV than unterraced ponds do. Nekton and SAV were not examined during winter.

Bird density was greater in terraced ponds than unterraced in both seasons. Bird species richness was generally greater in terraced ponds at most times during spring and summer, and was consistently greater in terraced ponds during winter. Bird density in both studies was positively correlated with the amount to of edge in ponds, and birds during spring and summer were observed to use edge habitat more than they used open water habitat. This was not evaluated for wintering birds.

Bird density in ponds varied by foraging guild in both studies. Dabbling foragers were denser in terraced ponds in both studies. Wading foragers were generally denser in terraced ponds during spring and summer, and were consistently denser during winter. Shorebirds and aerialist foragers were denser in terraced ponds during spring and summer, but did not differ in density between pond types during winter. When densities differed significantly between pond types, higher densities in terraced ponds probably result from the more abundant edges in these ponds providing an increased number of foraging sites. Additionally, the terraces themselves may be attractive as loafing areas.

Bird densities obtained for the two studies are intended to be relative measures of density between pond types rather than absolute densities (Table 13). Hurricane Rita struck the Chenier Plain during September of 2005, before winter sampling began. I have no pre-hurricane winter data, and damage to infrastructure necessitated a change in study methods. For this reason, it is impossible to attribute conclusive causes to changes in bird densities between studies. With this in mind, some general statements can be made. Two guilds, dabblers and shorebirds, are composed of mostly migratory species. These guilds were densest in seasons when more migrants are typically present, spring for shorebirds, and winter for dabblers. It is impossible to speculate about hurricane effects on shorebirds from this data. Low shorebird densities in winter were expected due to migration and high water levels in ponds. However, data for other guilds is suggestive. Winter migrants, such as dabbling ducks, were not present during September, and likely received no direct mortality from the hurricane. Winter migrants were also the densest species observed during winter. Both guilds of mostly resident birds, the waders and the aerialist foragers, were less dense during winter than during spring and summer. This may be a related to hurricane mortality or hurricane-caused dispersal. This is intriguing, but inconclusive. Differences in study methods alone could have generated this change. During spring and summer, seven bird counts were made per plot, and the maximum number of birds of a given species during any one count was used as the estimate of abundance. During winter, only one bird count was made per plot. Thus, based on chance, the probability of obtaining a higher density was greater for the spring and summer study.

Finally, results from my pilot study indicated that waterbird surveys in open water ponds using photosensitive camera systems were less efficient than were more traditional methods using observers.

Table 13. Raw mean bird density (birds/hectare) or species richness, +/- se, for the spring and summer study and the winter study. Methods varied between studies. It is unknown whether differences between studies resulted from seasonal differences, hurricane effects, or changes in study method. Asterisks indicate densities that did not differ significantly between pond types for any survey during that study.

	Spring and summer		Winter	
	Terraced	Unterraced	Terraced	Unterraced
Diver	0.08 (+/- 0.03)*	0.07 (+/- 0.03)*	0.08 (+/- 0.03)*	0.25 (+/- 0.08)*
Wader	0.5 (+/- 0.09)	0.2 (+/- 0.05)	0.17 (+/- 0.05)	0.04 (+/- 0.1)
Dabbler	0.4 (+/- 0.1)	0.2 (+/- 0.07)	4.2 (+/- 1.3)	0.9 (+/- 0.6)
Shorebird	1.2 (+/- 0.6)	0.05 (+/- 0.02)	0.14 (+/- 0.08)*	0.04 (+/- 0.04)*
Aerial	0.7 (+/- 0.2)	0.3 (+/- 0.15)	0.25 (+/- 0.08)*	0.2 (+/- 0.6)*
All birds	3.4 (+/- 0.9)	0.9 (+/- 0.2)	5.0 (+/- 1.3)	1.5 (+/- 0.2)
Species richness	6.7 (+/- 0.7)	4.6 (+/- 0.65)	2.7 (+/- 0.4)	1.3 (+/- 0.02)

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APPENDIX A: SCIENTIFIC NAMES OF BIRD SPECIES FOR ALL STUDIES

Table 14. Common and scientific name of bird species seen during spring and summer study.

Common Name	Scientific Name
Pied-billed Grebe	*Podilymbus podiceps*
Anhinga	*Anhinga anhinga*
American White Pelican	*Pelecanus erythrorhynchos*
Double-crested Cormorant	*Phalacrocorax auritus*
Neotropic Cormorant	*Phalacrocorax brasilianus*
American Bittern	*Botaurus lentiginosus*
Least Bittern	*Ixobrychus exilis*
Great Blue Heron	*Ardea herodias*
Great Egret	*Ardea alba*
Green Heron	*Butorides virescens*
Little Blue Heron	*Egretta caerulea*
Reddish Egret	*Egretta rufescens*
Snowy Egret	*Egretta thula*
Roseate Spoonbill	*Platalea ajaja*
Tricolored Heron	*Egretta tricolor*
Yellow-crowned Night-Heron	*Nyctanassa violacea*
Northern Shoveler	*Anas clypeata*
Fulvous Whistling-Duck	*Dendrocygna bicolor*
Gadwall	*Anas strepera*
Mallard	*Anas platyrhynchos*
Mottled Duck	*Anas fulvigula*
Common Moorhen	*Gallinula chloropus*
Purple Gallinule	*Porphyrio martinica*
King Rail	*Rallus elegans*
Clapper Rail	*Rallus longirostris*
Black-necked Stilt	*Himantopus mexicanus*
Black-bellied Plover	*Pluvialis squatarola*
Killdeer	*Charadrius vociferus*
Semipalmated Plover	*Charadrius semipalmatus*
Wilson Plover	*Charadrius wilsonia*
Greater Yellowlegs	*Tringa melanoleuca*
Long-billed Curlew	*Numenius americanus*
Lesser Yellowlegs	*Tringa flavipes*
Unknown sandpiper	*Calidris sp.*
Semipalmated Sandpiper	*Calidris pusilla*
Sharp-tailed Sandpiper	*Calidris acuminata*
Western Sandpiper	*Calidris mauri*

Willet	*Catoptrophorus semipalmatus*
Laughing Gull	*Larus atricilla*
Ring-billed Gull	*Larus delawarensis*
Black Tern	*Chlidonias niger*
Caspian Tern	*Sterna caspia*
Forster's Tern	*Sterna forsteri*
Gull-billed Tern	*Sterna niloctica*
Least Tern	*Sterna antillarum*
Royal Tern	*Sterna maxima*
Black Skimmer	*Rynchops niger*
Belted Kingfisher	*ceryle alcyon*

Table 15. Common and scientific names of bird species seen during the winter study.

Common Name	Scientific Name
Common Loon	*Gavia immer*
Eared Grebe	*Podiceps nigricollis*
Pied-billed Grebe	*Podilymbus podiceps*
Double-crested Cormorant	*Phalacrocorax auritus*
Neotropic Cormorant	*Phalacrocorax brasilianus*
Great Blue Heron	*Ardea herodias*
Great Egret	*Ardea alba*
Little Blue Heron	*Egretta caerulea*
Snowy Egret	*Egretta thula*
Tricolored Heron	*Egretta tricolor*
Roseate Spoonbill	*Platalea ajaja*
White Ibis	*Eudocimus albus*
Blue-winged Teal	*Anas dicors*
Gadwall	*Anas strepera*
Green-winged Teal	*Anas crecca*
Mallard	*Anas platyrhynchos*
Mottled Duck	*Anas fulvigula*
Northern Pintail	*Anas acuta*
Northen Shoveler	*Anas clypeata*
Red-breasted Merganser	*Mergus serrator*
Unknown duck	*Anas sp.*
Black-necked Stilt	*Himantopus mexicanus*
Killdeer	*Charadrius vociferus*
Greater Yellowlegs	*Tringa melanoleuca*
Lesser Yellowlegs	*Tringa flavipes*
Unknown sandpiper	*Calidris sp.*
Willet	*Catoptrophorus semipalmatus*
Laughing Gull	*Larus atricilla*
Ring-billed Gull	*Larus delawarensis*
Caspian Tern	*Sterna caspia*
Forster's Tern	*Sterna forsteri*
Gull-billed Tern	*Sterna niloctica*
Royal Tern	*Sterna maxima*

APPENDIX B: RAW BIRD DATA FOR ALL STUDIES

Table 16. Maximum bird density (birds/hectare) seen at any site for a given species, in terraced ponds during spring and summer 2005, Chenier Plain, LA. Parentheses have percent of plots sampled that day in which the species was observed.

Common Name	29-Apr	17-May	17-Jun	28-Jul	11-Aug	3-Sep
Pied-billed Grebe	0.08 (0.50)	0	0	0	0.22 (0.25)	0
Anhinga	0.08 (0.50)	0	0	0	0.29 (0.25)	0
American White Pelican	0	0	0	0	0	0
Double-crested Cormorant	0.08 (0.50)	0.47 (0.33)	0.29 (0.50)	0	0	0
Neotropic Cormorant	0	0	0	0	0	0
American Bittern	0	0	0	0	0.33 (0.50)	0
Least Bittern	0	0.36 (1.00)	0.31 (0.50)	0.33 (0.25)	0.17 (0.25)	0.08 (0.33)
Great Blue Heron	0	0.23 (0.67)	0.29 (0.50)	0.33 (0.50)	0.22 (0.25)	0
Great Egret	0.1 (0.50)	0.28 (0.33)	0.86 (0.50)	0.33 (0.50)	0.11 (0.50)	0
Green Heron	0	0	0.15 (0.50)	0	0	0
Little Blue Heron	0	0	0	0	0.33 (0.50)	0.29 (0.33)
Reddish Egret	0	0	0	0	0	0
Snowy Egret	0	0	0	0.1 (0.25)	0.29	0
Roseate Spoonbill	0.2 (0.50)	0	0	0	0	0
Tricolored Heron	0	0.47 (0.67)	0.29 (0.50)	0.23 (0.50)	0	0.29 (0.33)
Yellow-crowned Night-Heron	0	0	0.31 (0.50)	0	0	0.08 (0.33)
Northern Shoveler	0	0	0	0	0	0
Fulvous Whistling-Duck	0	0	0	0.23 (0.25)	0	0
Gadwall	0	0	0	0	0	0
Mallard	0	0.23 (0.33)	0	0	0	0
Mottled Duck	0	0.14 (0.33)	0	0	0	0

Common Moorhen	0.1 (0.50)	0.94 (1.00)	0.54 (1.00)	0.28 (0.75)	2.06 (0.50)	1.25 (0.33)
Purple Gallinule	0		0	0	0	0
King Rail	0	0.49 (0.33)	0	0	0	0
Clapper Rail	0	0	0	0	0.1 (0.25)	0
Black-necked Stilt	0	0.55 (1.00)	0.15 (0.50)	2.5 (0.50)	1.4 (0.25)	0.2 (0.33)
Black-bellied Plover	0	0	0	0	1.4 (0.25)	0.3 (0.33)
Killdeer	0	0.28 (0.33)	0	0	0.2 (0.25)	0.1 (0.33)
Semipalmated Plover	0	0	0	0.2 (0.25)	0	0
Wilson Plover	0	0	0	0	0	0
Greater Yellowlegs	0	0	0	0	0.1 (0.25)	0
Long-billed Curlew	0	0	0	0	0	0.1 (0.33)
Lesser Yellowlegs	0	0	0	6.5 (0.25)	0	0.1 (0.33)
Calidris sp.	0	0.55 (0.25)	0	7 (0.25)	1.1 (0.25)	0
Semipalmated Sandpiper	0	0	0	0	0.1 (.25)	0
Sharp-tailed Sandpiper	0	0	0	0	0.3 (.25)	0
Western Sandpiper	0	5.83 (0.67)	0	0	0	0
Willet	0	0.14 (0.33)	0	0.5 (0.25)	0.2 (0.25)	0.2 (0.33)
Laughing Gull	0	0.33 (0.33)	0	0.1 (0.50)	0.22 (0.50)	0
Ring-billed Gull	0	0.14 (0.33)	0	0	0	0
Black Tern	0	0.91 (0.67)	0.29 (0.50)	0.47 (0.25)	1.67 (0.25)	0.57 (0.33)
Caspian Tern	0.08 (0.50)	0	0.03 (0.50)	0	0	0
Forster's Tern	1.08 (1.00)	0	0.29 (1.00)	0.3 (0.25)	0.33 (0.25)	0
Gull-billed Tern	0	1.82 (0.33)	0.29 (0.50)	0.47 (0.25)	0.33 (0.75)	0.57 (0.33)
Least Tern	0	0.23 (0.67)	0.29 (0.50)	0	0	0
Royal Tern	0	0	0	0.1 (0.25)	0	0
Black Skimmer	0	0	0	0	0	0
Belted Kingfisher	0	0	0	0	0.33 (0.50)	0.29 (0.67)

Table 17. Maximum bird density (birds/hectare) seen at any site for a given species, in unterraced ponds during spring and summer 2005, Chenier Plain, LA. Parentheses have percent of plots sampled that day in which the species was observed.

Common Name	29-Apr	17-May	17-Jun	28-Jul	11-Aug	3-Sep
Pied-billed Grebe	0	0	0	0	0	0.25 (0.67)
Anhinga	0	0	0	0	0	0
American White Pelican	0	0	0	0	0	0
Double-crested Cormorant	0	0.17 (1.00)	0	0.08 (0.25)	0.17 (0.25)	0.25 (0.33)
Neotropic Cormorant	0	0	0.08 (0.50)	0	0	0
American Bittern	0	0	0	0	0	0
Least Bittern	0	0.09 (0.67)	0	0.11 (0.25)	0.22 (0.25)	0
Great Blue Heron	0	0	0	0.08 (0.25)	0.11 (0.25)	0.25 (1.00)
Great Egret	0	0.09 (1.33)	0	0.11 (0.25)	0	0.17 (0.67)
Green Heron	0	0	0	0	0.17 (0.25)	0
Little Blue Heron	0	0.08 (0.33)	0	0	0.33 (0.50)	0.17 (0.67)
Reddish Egret	0	0	0	0	0	0.25 (0.33)
Snowy Egret	0.17 (0.50)	0.08 (0.33)	0	0	0	0
Roseate Spoonbill	0	0	0	0	0	0.08 (0.67)
Tricolored Heron	0	0.08 (0.33)	0	0	0.08 (0.50)	0.08 (0.33)
Yellow-crowned Night-Heron	0	0	0	0	0	0
Northern Shoveler	0	0	0	0	0	0
Fulvous Whistling-Duck	0	0	0	0	0	0
Gadwall	0	0	0	0	0	0
Mallard	0	0	0	0	0	0
Mottled Duck	0	0.17 (0.33)	0	0	0.11 (0.25)	0.83 (0.33)
Common Moorhen	0.33 (0.50)	0.41 (1.00)	0	0	0.33 (0.50)	0.33 (0.67)
Purple Gallinule	0	0.08 (0.33)	0	0	0	0.08 (0.33)

King Rail	0	0	0	0	0.11 (0.25)	0
Clapper Rail	0	0	0	0	0	0.08 (0.33)
Black-necked Stilt	0.08 (0.50)	0	0	0	0	0.08 (0.67)
Black-bellied Plover	0	0	0	0	0	0
Killdeer	0	0	0	0	0	0
Semipalmated Plover	0	0	0	0	0	0
Wilson Plover	0	0	0	0.08 (0.25)	0	0
Greater Yellowlegs	0	0	0	0	0	0
Long-billed Curlew	0	0	0	0	0	0
Lesser Yellowlegs	0	0	0	0	0	0
Calidris sp.	0	0	0	0	0.25 (0.25)	0
Semipalmated Sandpiper	0	0	0	0	0	0
Sharp-tailed Sandpiper	0	0	0	0	0	0
Western Sandpiper	0	0	0	0	0	0
Willet	0	0.08 (0.33)	0	0	0.17 (0.25)	0.17 (0.33)
Laughing Gull	0	0.08 (0.33)	0	0.08 (0.25)	0.08 (0.25)	0
Ring-billed Gull	0	0.08 (0.33)	0	0	0	0
Black Tern	0	0.27 (0.33)	0.08 (0.50)	2.58 (0.75)	0.33 (0.25)	0.58 (0.33)
Caspian Tern	0	0	0	0.08 (0.25)	0	0.08 (0.33)
Forster's Tern	0.08 (0.50)	0	0	0.08 (0.25)	0.08 (0.25)	0.17 (0.33)
Gull-billed Tern	0	0.09 (0.33)	0	0.33 (0.50)	0.17 (0.50)	0
Least Tern	0	0.27 (0.33)	0	0	0	0
Royal Tern	0	0	0	0	0	0
Black Skimmer	0	0	0	0	0	0
Belted Kingfisher	0	0	0	0	0	0.08 (0.33)

Table 18. Maximum bird density (birds/hectare) seen at any site for a given species, in terraced ponds during winter 2006, Chenier Plain, LA. Parentheses have percent of plots sampled that day in which the species was observed.

Common Name	21-Jan	28-Jan	7-Feb	18-Feb	28-Feb	25-Mar
Common Loon	0	0	0	0	0	0
Eared Grebe	0.26 (0.13)	0	0	0	0	0
Pied-billed Grebe	0	0	0	0	0	0.43 (0.14)
Double-crested Cormorant	0.19 (0.25)	0	0.75 (0.17)	0	0	0.53 (0.29)
Neotropic Cormorant	0	0	0.56 (0.17)	0	0	0
Great Blue Heron	0.24 (0.25)	0	0.56 (0.17)	0.38 (0.17)	0.19 (0.20)	0.25 (0.14)
Great Egret	0.22 (0.13)	0	0.26 (0.33)	0.38 (0.33)	0.21 (0.20)	0.38 (0.14)
Little Blue Heron	0	0.22 (0.25)	0.19 (0.17)	0	0	0
Snowy Egret	0	0	0	0	0	0
Tricolored Heron	0	0	0	0.19 (0.17)	0.21 (0.20)	0
Roseate Spoonbill	0	0	0	0.88 (0.17)	0	0
Blue-winged Teal	1.1 (0.13)	0	0.88 (0.17)	0	0.64 (0.40)	4.26 (0.29)
Gadwall	1.51 (0.38)	1.53 (0.25)	3.57 (0.50)	0	0	7.45 (0.29)
Green-winged Teal	0.79 (0.13)	0	12.42 (0.33)	17.54 (0.33)	10.65 (0.40)	32.45 (0.57)
Mallard	0	0	1.32 (0.33)	0	0	0
Mottled Duck	0	0.51 (0.25)	0	0	0	0.42 (0.14)
Northern Pintail	1.85 (0.13)	7.93 (0.25)	0.88 (0.50)	0	0	0
Northern Shoveler	0	0	0.63 (0.33)	0	3.19 (0.20)	3.44 (0.14)
Unknown Duck Sp	0	0	2.19 (0.17)	0	2.63 (0.20)	0
Red-breasted Merganser	0	0	0	0	0	0
Black-necked Stilt	0	0	0	0	0	0.25 (0.14)
Killdeer	0.44 (0.50)	0	0.38 (0.17)	0	0	0
Greater Yellowlegs	0	0	0	0	0	0.76 (0.29)

Calidris sp.	0	0	0	0	0.64 (0.20)	0
Lesser Yellowlegs	0	0	0	0	0	0
Whimbrel	0	0	0.88 (0.17)	0	0	0
Willet	0	0	0	0	0	2.28 (0.14)
Laughing Gull	0.38 (0.13)	0	0	0.38 (0.17)	0	0
Ring-billed Gull	0	0	0	0.19 (0.17)	0.43 (0.20)	0
Caspian Tern	0.22 (0.13)	0	0.25 (0.33)	0	0.22 (0.20)	0
Forster's Tern	0.19 (0.13)	0.51 (1.00)	2.38 (0.17)	0	0.42 (0.20)	0.88 (0.14)
Gull-billed Tern	0.19 (0.13)	0	0	0	0	0
Royal Tern	0	0.22 (0.50)	0.19 (0.17)	0	0.21 (0.20)	0

Table 19. Maximum bird density (birds/hectare) seen at any site for a given species, in unterraced ponds during winter 2006, Chenier Plain, LA. Parentheses have percent of plots sampled that day in which the species was observed.

Common Name	21-Jan	28-Jan	7-Feb	18-Feb	28-Feb	25-Mar
Common Loon	0.19 (0.2)	0	0	0.19 (0.17)	0	0
Eared Grebe	0	0	0	0	0	0
Pied-billed Grebe	0.57 (0.20)	0.22 (0.33)	0.23 (0.17)	0.19 (0.17)	0	0
Double-crested Cormorant	0.23 (0.20)	0.23 (0.33)	4.1 (0.33)	0.23 (0.17)	0	0.22 (0.14)
Neotropic Cormorant	0	0	0	0	0	0
Great Blue Heron	0	0	0	0	0	0
Great Egret	0.38 (0.40)	0.19 (0.17)	0	0.19 (0.17)	0	0.33 (0.14)
Little Blue Heron	0	0	0	0	0	0
Snowy Egret	0	0.22 (0.17)	0	0	0	0
Tricolored Heron	0	0	0	0	0	0
Roseate Spoonbill	0	0	0	0	0	0
Blue-winged Teal	0	0	0	0	0	3.21
Gadwall	0	0.23 (0.17)	0.19 (0.17)	3.09 (0.17)	0.68 (0.50)	15.9 (0.14)
Green-winged Teal	0	0	0	0	0	3.41 (0.14)
Mallard	0	0	0	0	0	0
Mottled Duck	0	0	0	0	0.44 (0.25)	0.54 (0.14)
Northern Pintail	0	0	0	0	0	0
Northern Shoveler	0	0	0	0	0	3.8 (0.14)
Unknown Duck	0	0	0	0	0	0
Red-breasted Merganser	0	0	0.67 (0.17)	0	0	0
Black-necked Stilt	0	0	0	0	0	0
Killdeer	0	0	0	0	0	0
Greater Yellowlegs	0	0	0	0	0	0.89 (0.14)

Calidris sp.	0	0	0	0	0	0
Lesser Yellowlegs	0	0	0	0	0	0.54 (0.14)
Whimbrel	0	0	0	0	0	0
Willet	0	0	0	0	0	0
Laughing Gull	0	0	0.22 (0.17)	0	0	0
Ring-billed Gull	0.44 (0.20)	0	0.31 (0.17)	0.44 (0.17)	0.34 (0.25)	0
Caspian Tern	0	0	0	0	0	0
Forster's Tern	1.07 (0.40)	0.18 (0.17)	2.14 (0.17)	0	0	0.45 (0.29)
Gull-billed Tern	0	0	0	0	0	0
Royal Tern	0	0	0	0	0	0

APPENDIX C: NEKTON BIOMASS IN COASTAL LOUISIANA DURING SPRING AND SUMMER

This appendix contains an alternate analysis for the nekton samples collected during the spring and summer study. For a detailed introduction, please see Chapter 2. In that chapter, nekton density was discussed. Nekton biomass may also differ between pond types and microhabitat types, and is another equally valid analysis for these data.

METHODS

Collection Methods

For detailed collection methods, please see Chapter 2.

Statistical Methods

All statistical analyses were conducted using SAS 9.1.2 (SAS Institute Inc., 100 SAS Campus Drive, Cary, NC 27513-2414, USA). Multiple days were required to sample all sites. However, for the purpose of analysis, I assigned each survey a single date (the average of the dates over which the survey took place). For analysis, I labeled the second Vermilion Parish pond pair as plots within the Vermilion Parish site because it was not hydrologically distinct from the other ponds sampled in Vermilion Parish. Thus, on surveys where both pairs of Vermilion ponds were sampled, they were analyzed as day-site replicates of each other.

I used a repeated measures ANOVA with blocking on site to compare nekton biomass (g/m^2) between pond types (terraced or unterraced). I additionally included the microhabitat (open or edge) in which the samples were collected as an independent factor in the model (Chapter 2, Table 3). Two *a priori* contrasts were analyzed, comparing nekton biomass at marsh edge in terraced and unterraced ponds, and comparing nekton biomass in open water in terraced and unterraced ponds. Response variables were log transformed to achieve normality and reduce heterogeneity of variances.

RESULTS

Whole-Pond Analysis

Adding terraces to ponds did not significantly increase nekton biomass ($F_{1,14}$ = 1.55 p = 0.23) at the whole-pond scale. Mean log of nekton biomass (g/m^2)was 1.58, se = 0.47, (raw average = 19.6, se = 3.9) in terraced ponds, and 2.0, se = 0.47, (raw average =39.6, se = 3.6) in unterraced ponds.

Microhabitat Analysis

Terraced ponds and unterraced ponds had similar nekton biomass (g/m^2) at the marsh edge (Figure 26, $F_{1,32}$ = 2.03, p = 0.16). Similarly, the two pond types have similar nekton biomass in open water (Figure 26, $F_{1,32}$ = 2.38, p= 0.13). When data from terraced and unterraced ponds are combined, nekton biomass ($F_{1,29}$ = 4.01 p = 0.0548) did not differ between microhabitat types, though a non-significant trend was evident. Mean log of nekton biomass g/m^2 was 1.55, se = 0.47, (raw average = 25.39, se = 2.8) in open water habitat, but was 2.03, se = 0.47, (raw average = 21.82, se = 2.7) in marsh edge habitats.

DISCUSSION AND CONCLUSION

The results for nekton biomass are similar to those seen for nekton density (see chapter 2). The magnitude of difference between microhabitat types is not as strong for nekton biomass as it was for nekton density, but the overall trend is similar. Thus, the discussion and conclusions provided for nekton density in chapter 2 also applies to nekton biomass.

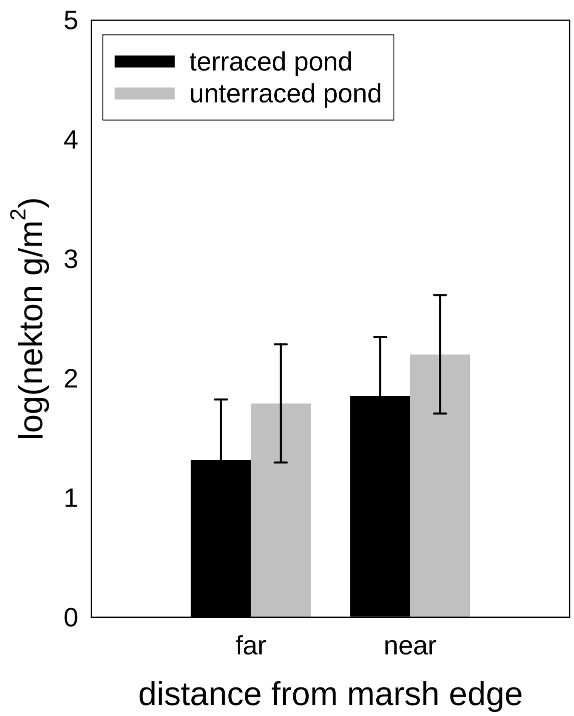

Figure 26. Nekton biomass in terraced and unterraced ponds at two microhabitat types, in spring and summer of 2005, Chenier Plain, Louisiana.

VITA

Jessica Leighton O'Connell was born in Wiesbaden, Germany, and grew up in Sonoma County, California. She received an Associate of Science from Santa Rosa Junior College (2000) and a Bachelor of Arts from Sonoma State University (2003), both in biology. She began her wildlife career as a wild bird rehabilitator at the Santa Rosa Bird Rescue Center in California. Following graduation, she worked for PRBO Conservation Science as an intern monitoring Song Sparrows in the salt marshes surrounding San Francisco Bay, California. Additionally, she worked as a PRBO staff biologist, monitoring Brown Pelican and other seabird populations at Vandenberg Air Force Base, California. Following this, she worked for Dr. Sammy King of Louisiana State University, as a field technician performing various wildlife surveys. She began her Master of Science at Louisiana State University under Dr. J. Andrew Nyman in fall of 2004. Following graduation, she will pursue a Doctor of Philosophy at Tulane University under the direction of Dr. Tom Sherry.